Aliens and humans: an ecosystem services perspective on plant invasions in the Anthropocene

Ana Sofia Vaz

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Department of Biology, Faculty of Sciences, University of Porto
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Supervisor
João P. Honrado, Professor at Faculty of Sciences, University of Porto

Co-supervisors
Domingo Alcaraz-Segura, Professor at Department of Botany, University of Granada

Joana R. Vicente, Post-Doctoral Researcher at Research Network in Biodiversity and Evolutionary Biology, Research Centre in Biodiversity and Genetic Resources

Ao meu pai.

FOREWORD / NOTA PRÉVIA

According to the General Regulation of Doctoral Programs of the University of Porto (number 2, 4th Article) and the Decree Law 74/2006 (Article 31st, 24th of March) revised under the Decree law 230/2009 (14th of September), this thesis includes manuscripts published or in consideration for publication in peer-review scientific journals. These manuscripts result from collaborations with several co-authors. The candidate declares that she actively contributed to the ideas and the development of the research work, including the compilation, analysis, results, discussion and writing as in its current publication form. The candidate was supported by the National Foundation for Science and technology (FCT), through a PhD Grant (PD/BD/52600/2014) financed by the European Social Fund and by the National Ministry of Science, Technology and Higher Education, through the Operational Programme Human Capital (POCH), under Portugal 2020. This thesis' research was developed in the context of the Doctoral Programme in Biodiversity, Genetics and Evolution (Faculty of Sciences), which started in September 2014. After a one-year period of classes it ended in August 2018.

Na elaboração desta tese, e nos termos do número 2 do Artigo 4º do Regulamento Geral dos Terceiros Ciclos de Estudos da Universidade do Porto e do Artigo 31o do D.L. 74/2006, de 24 de março, com a nova redação introduzida pelo D.L. 230/2009, de 14 de setembro, foi efetuado o aproveitamento de um conjunto coerente de trabalhos de investigaçãó publicados ou submetidos para publicação em revistas internacionais indexadas e com arbitragem científica, os quais integram alguns dos capítulos da presente tese. Os referidos trabalhos foram realizados com a colaboração de outros coautores. O candidato declara que participou ativamente na conceção, na obtenção, análise e discussão de resultados, e na elaboração da forma publicada destes trabalhos. O candidato foi apoiado pela Fundação para a Ciência e Tecnologia (FCT) através de bolsa de doutoramento (PD/BD/52600/2014) cofinanciada pelo Fundo Social Europeu e pelo Ministério da Ciência, Tecnologia e Ensino Superior, através do Programa Operacional Capital Humano (POCH), do Portugal 2020. O trabalho desta tese foi desenvolvido no contexto do plano doutoral em Biodiversidade Genética e Evolução (Faculdade de Ciências), que se iniciou em setembro de 2014. Após um ano curricular, terminou em agosto de 2018.

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This section traditionally starts with a formal recognition of the supervisors' support. Then, it acknowledges the team-work efforts from the colleagues. Finally, it ends with an expression of gratitude toward all supportive friends and family members. However, I decided not to follow this structure, since many of my colleagues are my friends, my friends constitute my family, and my family…well, it's whoever I want it to be!

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If I missed someone (I'm sure I did), please mind that I just wrote a PhD thesis…/ Se eu me esqueci de alguém (com certeza que sim), por favor tenham em atenção que eu acabei de escrever uma tese de doutoramento…

Ao Max, pela carinhosa e fiel companhia durante estes últimos meses!

ABSTRACT

The large-scale redistribution of species worldwide is a key fingerprint of the Anthropocene. Alien species, and particularly invasive species, challenge many issues pertaining the conservation of biodiversity as well as the functioning of ecosystems. These challenges bring consequences for the services that ecosystems sustain, and hence benefits and nuisances that people use, value and perceive. The benefits and nuisances related to biological invasions are influenced by human perception, exposure and action, depending on the ecological changes that these species cause at particular socio-cultural contexts. Therefore, combining different disciplinary views and scientific methods, from ecological and environmental sciences to social sciences and humanities, can aid in the understanding of invasions as social-ecological phenomena. Concurrently, by taking advantage of the technological advances of remote sensing in invasion science, the ability to deal with alien species and their social-ecological changes can be improved.

This thesis aims to contribute to the understanding and management of biological invasions as a social-ecological phenomenon, following and combining two major lines of investigation: social-ecological approaches and remote sensing. Seven studies are developed, aiming to answer the following questions: (1) Can an ecosystem (dis)services framework, grounded on the benefits and nuisances for human well-being, improve the understanding and management of biological invasions as a social-ecological phenomenon? (2) How have invasion research and management taken advantage of the opportunities provided by remote sensing advances, and how can they further benefit from those opportunities? and (3) Can social-ecological approaches and remote sensing be combined in integrative frameworks that effectively improve the future assessment and management of invasions?

Following a general introduction to the research context and objectives, the first study proposes a conceptual framework to integrate ecosystem services and disservices for human well-being. The framework is illustrated considering the multiple benefits and nuisances from plant invasions worldwide. It also accounts for the role of social-ecological management in the valuation of ecosystem services and disservices associated to plant invasions. This study proposes a precautionary management hierarchy to be used when considering ecosystem services and disservices promoted by invasions.

Subsequently, two studies are presented focusing on the usefulness of social-ecological approaches in addressing alien species from an ecosystem services perspective. These social-oriented approaches are first reviewed, providing an overview on how to foster collaboration and insights from social sciences to understand the role of biological invasions as promoters or disrupters of ecosystem services. The usefulness of social-ecological approaches is then illustrated in a case-study focused on assessing the effects of alien trees on multiple cultural services in the Iberian Peninsula, from the analysis of social media data.

The next two studies focus on the opportunities offered by remote sensing to invasion science. Firstly, the progress, state and applications of remote sensing in the research and management of plant invasions are reviewed and discussed. Afterwards, remote sensing advances are combined with social-ecological approaches to evaluate the seasonal contribution of alien tree species to cultural services in a Portuguese National Park. Recommendations are proposed for informed management regarding alien trees, focused on the safeguard of natural capital and recreational benefits.

Building on the previous social-ecological and remote sensing approaches, the last two studies propose potential ways forward in the research of ecosystem services and in the science of biological invasions. First discussing the many opportunities that remote sensing offers for the understanding of cultural ecosystem services at a planetary scale, and then arguing the consideration of remote sensing as an emerging issue of increasing relevance and applicability in invasion science.

Finally, the implications of the different studies are discussed for advancing invasion science from an ecosystem services perspective. Overall, the benefits or nuisances deriving from alien species are defined by human perception, exposure, and valuation at particular geographical, temporal and socio-cultural settings. Nevertheless, alien species inevitably shape ecosystem attributes, process and functions, which management can benefit from the many opportunities offered by remote sensing. Understanding and managing biological invasions as a social-ecological phenomenon should consider interdisciplinary efforts that combine research cultures, questions and methods from social-oriented disciplines and technological advances.

The conclusions of this thesis settle on advancing the thinking on ecosystem (dis)services and acknowledging the pivotal role of human perceptions, actions and technology on improving the management of invasions in the Anthropocene.

Keywords: Biodiversity conservation; Biological invasions; Earth observation; Human well-being; Iberian Peninsula; Nature Contributions to People; Non-native trees; Protected areas; Remote sensing; Research synthesis; Social-ecology.

RESUMO

A redistribuição de espécies à larga escala mundial constitui uma impressão digital chave do Antropoceno. As espécies exóticas, e particularmente as espécies invasoras, colocam muitos desafios à conservação da biodiversidade e ao funcionamento dos ecossistemas. Estes desafios trazem consequências para os serviços que os ecossistemas suportam, e consequentes benefícios que as pessoas usam, valorizam e percecionam. Os benefícios e as limitações impostas pelas espécies invasoras são influenciados pela perceção e ação humana, e dependem das alterações ecológicas que estas espécies induzem em contextos particulares socioculturais. Desta forma, a combinação de diferentes perspetivas e métodos disciplinares, incluindo aqueles derivados das ciências ambientais, sociais e humanísticas, pode ajudar a clarificar o papel das invasões biológicas como um fenómeno socio-ecológico. Simultaneamente, tirar partido dos crescentes avanços tecnológicos inerentes à deteção remota, pode constituir uma oportunidade para melhor lidar com as espécies exóticas e respetivos impactos.

Esta tese pretende contribuir para a compreensão e gestão das invasões biológicas como um fenómeno socio-ecológico, seguindo e combinando duas linhas principais de investigação: abordagens socio-ecológicas e deteção remota. Para isso, sete estudos são apresentados, com o intuito de responder às seguintes questões: (1) Pode uma moldura de (dis)serviços dos ecossistemas, baseada nos benefícios e limitações ao bem-estar humano, contribuir para a compreensão e gestão das invasões biológicas como um fenómeno socio-ecológico? (2) Como têm a investigação e a gestão das invasões biológicas aproveitado as oportunidades associadas aos avanções da deteção remota, e como podem vir a ser mais beneficiados por essas oportunidades? e (3) Podem as abordagens socio-ecológicas e a deteção remota ser combinadas em molduras integradoras que melhorem a avaliação e a gestão das invasões no futuro?

Seguindo uma introdução geral sobre o contexto e objetivos de investigação, o primeiro estudo apresenta uma moldura conceptual para integrar os serviços e disserviços dos ecossistemas para o bem-estar humano. A moldura é ilustrada considerando os múltiplos benefícios e limitações produzidos pelas invasões biológicas a nível mundial. A moldura considera também o papel da gestão socio-ecológica e da valoração dos serviços e disserviços dos ecossistemas associados às invasões biológicas. O estudo propõe uma

hierarquia de gestão precautória a ser usada aquando a consideração dos serviços e disserviços dos ecossistemas promovidos pelas invasões.

Os dois estudos que se seguem são focados na utilidade das perspetivas socio-ecológicas sobre as espécies exóticas segundo uma perspetiva de serviços dos ecossistemas. Estas abordagens socio-ecológicas são primariamente revistas, oferecendo uma visão global de como promover a colaboração e o conhecimento com as ciências sociais por forma a compreender o papel das invasões biológicas como impulsionadoras ou disruptoras dos serviços dos ecossistemas. De seguida, um caso de estudo focado na aplicação de abordagens socio-ecológicas é adotado para analisar os efeitos das árvores exóticas em vários serviços culturais na Península Ibérica, a partir de dados de média sociais.

Os dois estudos subsequentes são focados nas oportunidades oferecidas pela deteção remota na ciência da invasão. Primariamente, o progresso, estado e aplicações da deteção remota são revistos e discutidos no contexto da investigação e gestão das plantas invasoras. De seguida, os avanços da deteção remota são combinados com abordagens socio-ecológicas para avaliar a contribuição sazonal das árvores exóticas nos serviços dos ecossistemas culturais num Parque Nacional de Portugal. Recomendações para a gestão de árvores exóticas são propostas com o objetivo de salvaguardar o capital natural e cultural.

Com base nas abordagens socio-ecológicas e nos avanços inerentes à deteção remota, os últimos dois estudos propõem caminhos potenciais a seguir na investigação dos serviços dos ecossistemas e na ciência das invasões biológicas. Primeiramente, as oportunidades que a deteção remota oferece para compreender os serviços dos ecossistemas culturais são discutidas à escala planetária. De seguida, a deteção remota é apresentada como um assunto emergente e de crescente relevância e aplicabilidade na ciência da invasão.

Finalmente, as implicações dos diferentes estudos são discutidas por forma a avançar a ciência da invasão através de uma perspetiva de serviços dos ecossistemas. Em síntese, os benefícios e limitações derivados das espécies exóticas são definidos pela exposição e perceção humanos, dependendo da valoração que as pessoas fazem em determinados contextos geográficos, temporais e socioculturais. No entanto, as espécies exóticas (e invasoras) inevitavelmente moldam os atributos, processos e funções dos ecossistemas, para os quais a gestão pode beneficiar das oportunidades da deteção remota. Neste sentido, compreender e gerir as invasões biológicas como um fenómeno socio-ecológico deverá considerar esforços que combinem culturas, questões e métodos de investigação derivados das disciplinas sociais e de avanços tecnológicos.

As conclusões desta tese assentam no avanço do pensamento sobre os (dis)serviços dos ecossistemas e no reconhecimento do papel fundamental da perceção, ação e tecnologia humanas na gestão das invasões biológicas no Antropoceno.

Palavras-chave: Áreas protegidas; Árvores não-nativas; Bem-estar humano; Ciência de síntese; Conservação da biodiversidade; Contribuições da Natureza para as Pessoas; Invasões biológicas; Deteção remota; Investigação de Síntese; Península Ibérica; Observação da Terra; Socio-ecologia.

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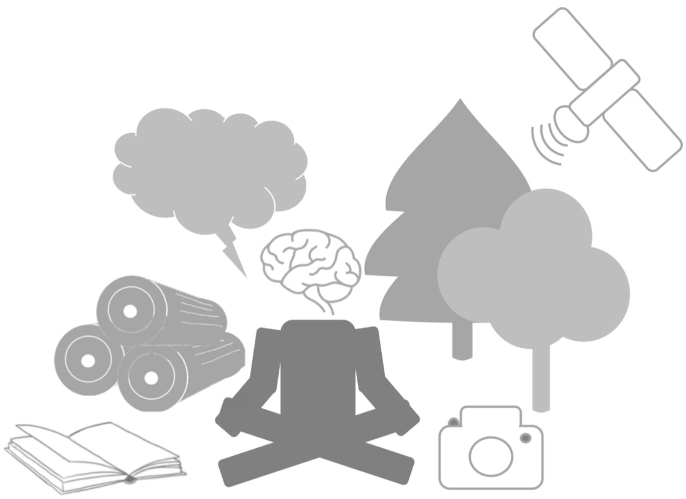
CHAPTER 1. INTRODUCTION



1.1. From the Neolithic to modern times: the homogenisation of biota

For millennia, mankind has been responsible for the transportation and introduction of a variety of alien species to other locations for cultural and subsistence reasons. Approximately 10 000 years ago, the rise of agriculture fostered the movement of edible plants and useful animals, which, together with the domestication of local species and varieties, ensured the start of the sedentary *Homo sapiens sapiens* (Mazoyer and Roudart, 2007). From this Neolithic revolution onwards, human management has progressed based on techniques that allowed to take advantage of native and alien species (see Box 1) which could "serve" people's daily needs and promote human well-being, namely through food (e.g. crops and meat), shelter (e.g. wood and fur) and protection (e.g. shrubs and reeds for fences).

The transportation of species has increased during the times that followed. A significant movement of alien species can be recognised during the Roman period, a time when technological advances and the surge of commercialisation brought together wide territories, such as the Orient, Northern Africa, and Europe. Nevertheless, it was the start of the maritime expansions by Europeans in the 15th century that connected continents through the oceans, leading to an unprecedented exchange of species at the global scale (Crosby, 2004). During the subsequent centuries, the discovery of new species and respective novel uses, the admiration of "out-of-normal" and "exotic" biota, and the return of living beings as colonial evidence by European empires, motivated the start of a biological globalisation (Murphy, 2007; Pooley, 2018; Rotherham, 2011).

During the 18th and 19th centuries, the emergence of physiocracy and acclimatisation[1] led to the intensification of this biological globalisation (Osborne, 2000). Many alien species were traded for horticulture, aesthetics, collecting, gardening, and for developing more productive farming and forestry systems. It was during this time that transatlantic trades and "philosophical travels" accelerated, therefore promoting the (intentional and accidental) introduction of alien species which were already cultivated and often re-distributed for several purposes (Pooley, 2018). This was especially relevant to meet the increasing economic and demographic needs from the start of industrialisation and urbanisation ages.

[1] Physiocracy is one of the firsts economic theories, emerging in the 18th century, that was grounded on the believe that the wealth of nations derived from the value of land development and that agricultural products should be highly priced. Acclimatisation emerges in the 19th century by voluntary associations to encourage the introduction of alien species, with the expectation of their acclimatisation and adaptation, so as to enrich the value of the land (Osborne, 2000).

Box 1: The (introduction-)naturalization-invasion continuum

There are several definitions which can be used to address biological invasions. This thesis adopts the terminology from Richardson et al. (2000, 2011), Richardson and Pyšek (2006) and Essl et al. (2018):

Alien / non-native: species that are introduced, accidentally or intentionally, by humans to new geographic areas (synonyms: non-indigenous, allochthonous, exotic). Plant species introductions before or after the "Columbian exchange" (year 1492) are considered as "archeophytes" or "neophytes", respectively;

Casual: alien species that may reproduce occasionally in an area, but that do not form self-replacing populations, and that rely on human agency and repeated introductions for their persistence;

Naturalised: alien species that reproduce consistently and sustain populations over many life cycles without direct human intervention/agency; they often recruit offspring freely, usually close to adult plants, and do not necessarily invade ecosystems;

Invasive: naturalised species that spread over considerable areas, sometimes becoming abundant and leading to major impacts on the environment and society. The status of invasive species can be considered based on their origin, time since introduction/residence time, and degree of invasion.

This terminology relates to different stages of the invasion process. This process takes place when species are able to cross a set of geographical, (socio-)environmental and physiological barriers to become alien, casual, naturalised or invasive in new areas (see Figure B1). These barriers occur at different scales (e.g. regional, local or landscape), forming the so-called (introduction-)naturalization-invasion continuum.

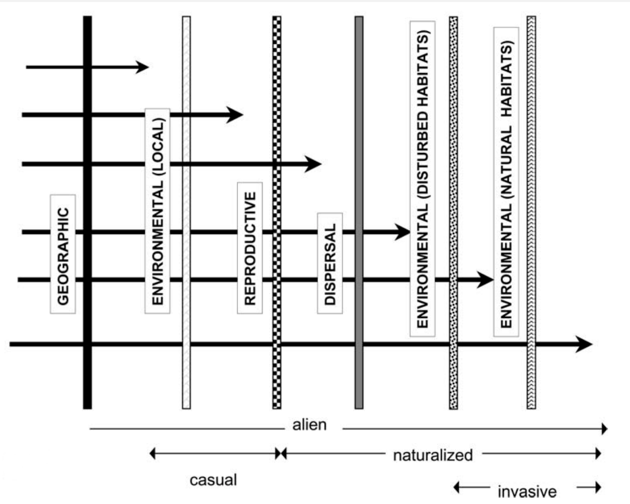


Figure B1. Representation of the (introduction-)naturalization-invasion continuum, showing the barriers that a species crosses to become alien, casual, naturalised or invasive in a new area (from Richardson and Pyšek, 2006).

In this continuum, the first barriers are mostly geographic, and depend on human behaviours to introduce and establish species (Essl et al., 2018). The other barriers relate to environmental conditions, and to the capacity of the species to cope with such conditions. Species are able to move forward on the (introduction)naturalization-invasion continuum depending on the susceptibility of the new area to species invasions (invasibility) as well as on the traits of the alien species which allow them to invade the new area (invasiveness; Richardson and Pyšek, 2006). Time lags may occur until which the appropriate environmental conditions are created, or alien species become adapted to the new area (invasion lag; Crooks, 2005).

From the many alien species introduced deliberately, accidently or by association, a few escaped from cultivated and confined areas and became invasive (see Box 1), often producing economic or environmental impacts (Ritvo, 2014; Rotherham, 2011). Nevertheless, it was in the apogee of modern globalisation, expressed by dynamic economic, political and demographic changes during the 20th century (i.e. "the great acceleration"; Steffen et al., 2015), that invasive species became recognised as a major threat to the conservation of biodiversity and to the sustainable use of natural resources (Drake et al., 1989). The emergence of many pests and diseases associated to introduced alien species (e.g. as recognised by the *International Plant Protection Convention* in 1951) has paved the way for the recognition of biological invasions as a global environmental concern of anthropogenic origin (McNeely, 2001). In the current era of changing climates and land cover uses, as well as of dynamic human-environment interactions, the phenomenon of biological invasions became a key fingerprint of the Anthropocene (Kueffer, 2017).

1.2. BEAUTY OR BEAST? ALIEN SPECIES IN THE ANTHROPOCENE

The Anthropocene, or the human epoch, is expressed by the dominance of global scale human activities, which constitute the main cause of (most) contemporary environmental changes (Lewis and Maslin, 2015). Therefore, the Anthropocene is characterised by unprecedented levels of anthropogenic change that push the planetary boundary conditions beyond their safe operating spaces[2] (Bennett et al., 2016; Rockström et al., 2009). Alongside climate and land use changes, pollution, resource overexploitation and other challenges, alien species, and particularly biological invasions, prevail amongst the most serious fingerprints of the Anthropocene (Head et al., 2015; Hui and Richardson, 2017; Kueffer, 2017).

The intentional introduction of alien species has been widely motivated to improve natural capital and to promote the supply of ecosystem services (see Box 2), such as wood production, soil stabilisation and landscape aesthetics (Kueffer and Kull, 2017; Kull et al., 2011; Vaz et al., 2017b). In the increasingly globalised world of the Anthropocene, alien species are still key resources in many regions worldwide, supporting daily basic needs of local communities. Examples include the plantation of *Eucalyptus globulus* for the pulp industry and for the improvement of national economy in Portugal (Krumm and Vítková, 2016),

[2] In short, planetary boundaries express the limits within which humans can modify Earth's biophysical subsystems or processes (i.e. the safe operating space) without crossing critical threshold values from which those subsystems and processes shift to a new state, often with deleterious or potentially even disastrous consequences for humans (Rockström et al., 2009).

or the trade of *Opuntia ficus-indica* fruits for financial sustenance in South Africa (Shackleton et al., 2007).

Box 2: From ecosystem services to people's well-being

The term *ecosystem services* has been coined (Ehrlich and Ehrlich, 1981) to refer to the utilitarian nature of ecosystem functions that are used and perceived as human benefits. Ecosystem services have been linked to other concepts such as "eco-services" (Costanza et al., 2014), "nature-based solutions" (Potschin et al., 2015) or "nature's contributions to people" (Díaz et al., 2018).

Emblematic initiatives targeting ecosystem services include the *Millennium Ecosystem Assessment* (MEA, 2005), the *Ecosystem Services Partnership* (ESP 2008), *The Economics of Ecosystems and Biodiversity* (TEEB, 2010), the *Mapping and Assessment of Ecosystems and their Services* (MAES, 2013), the *Common International Classification of Ecosystem Services* (CICES; Haines-Young and Potschin, 2018), and the *Intergovernmental Platform on Biodiversity and Ecosystem Services* (Díaz et al., 2015). These initiatives consider slightly different definitions and typologies of ecosystem services. For simplicity, this thesis considers three main categories of ecosystem services (Figure B2):

Provisioning: as the material products obtained from ecosystems, including food, timber, and energy;
Regulating (and maintenance): as the benefits derived from the regulation of ecosystem processes such as natural hazard regulation, water purification and waste management;
Cultural: non-material benefits obtained from ecosystems, namely through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences.

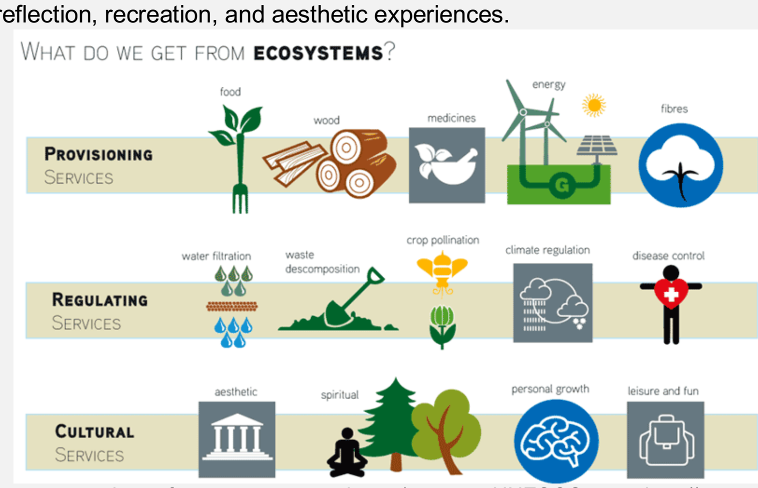


Figure B2. The three categories of ecosystem services (source: UNESCO, at: http://www.ehu.eus/cdsea/web/wp-content/uploads/2016/11/EcosystemServices.gif).

The concept of ecosystem services is an opportunity for considering natural capital in policy and management, and for demonstrating how global phenomena of social-ecological change, such as alien invasive species, affect ecosystem functions. These functions are necessary to support human well-being (Costanza and Daly, 1992; Gómez-Baggethun et al., 2010), which results from (a) objective attributes related to people's material and social contexts, namely physical materials, employment, education and health; (b) subjective individual's thoughts, feelings and satisfactions towards life's circumstances, such as perception of beauty and pleasure, or sense of fear and place; and (c) psychological responses, such as social connectedness, security, and life satisfaction (King et al., 2014; Smith et al., 2013; Summers et al., 2012).

Nevertheless, alien invasive species also cause undesirable impacts[3], by altering ecosystem processes, reducing native biodiversity and competing with other service-providing species (Levine et al., 2003; Simberloff et al., 2013). These impacts can reduce the supply of ecosystem services and therefore affect human well-being (Vaz et al., 2017b; Vilà and Hulme, 2017). Notable examples include the depletion of water for human consumption by invasive *Prosopis* species (Carruthers et al., 2011), and the reduction of human safety through the amplification of fire regimes by introduced *Acacia* species (Gaertner et al., 2014; Potgieter et al., 2018). Also, alien species can lead to direct nuisances for human well-being, by promoting ecosystem disservices (Shackleton et al., 2016; Vaz et al., 2017b), such as physical injuries caused by contact with *Opuntia* thorns (Shackleton et al., 2007) or pollen allergy and dermatitis associated to *Cortaderia selloana* (Pyšek and Richardson, 2010), among other examples (Vaz et al., 2017b).

Occasionally, the impacts of alien invasive species can be so severe that they can act as ecosystem engineers and keystone species, promoting regime shifts and novel ecosystems[4] with major social, economic and cultural implications (Gaertner et al., 2014; Kull et al., 2018; Simberloff, 2015). Therefore, there are many challenges, synergies and trade-offs (see Box 3) among beneficial and detrimental impacts of biological invasions to people (Essl et al., 2017; Humair et al., 2014; van Wilgen and Richardson, 2014) which change across spatial, temporal and social contexts (Kull et al., 2018; Shackleton et al., 2018a; Shackleton et al., 2018b). Consequently, in many situations, pursuing nature conservation alongside human well-being in the light of invasions might be only possible at the social-ecological interface and within an interdisciplinary arena (Kueffer, 2013; Kueffer and Hadorn, 2008; Shackleton et al., 2018b; Vaz et al., 2017a).

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[3] Although knowing there are slightly different definitions, this thesis uses the term “impact” as a synonym of “effect” to refer to a change promoted by alien species (Jeschke et al., 2014). regardless of the direction of the impact (positive or negative), unless otherwise indicated.

[4] The concept of ecosystem engineer refers to *“a species that alters resource availability to other species through nontrophic behaviours or structures”*, whereas keystone species expresses a *“species that has an effect on other species and/or on material fluxes out of proportion to its abundance, and through entirely biotic mechanisms”* (Ehrenfeld, 2010). The changes produced by engineer and keystone alien species may lead to regime shifts, i.e. profound changes in one or more processes that sustain the dynamic patterns and conditions that characterise and maintain a particular regime (Biggs et al., 2012; Kull et al., 2018), leading to a new ecosystem (i.e. a novel ecosystem; Simberloff, 2015).

1.3. ALL EYES ON ALIENS: INTERDISCIPLINARITY FOR EFFECTIVE MANAGEMENT

1.3.1. The study of biological invasions

Charles Elton's book[5] on *The Ecology of Invasions by Animals and Plants* (Elton, 1958) attempted to bring together different disciplines, including ecology, evolution, biogeography, biological conservation, and social sciences. Nevertheless, the study of biological invasions has been rooted in natural sciences, particularly in ecology and environmental disciplines (Richardson and Pysek, 2008). The biology and ecology of alien and invasive species has, for several decades, focused on issues related to forestry, agricultural pests, fish and game management, livestock diseases, and wildlife conservation (Davis et al., 2001; Lockwood et al., 2007). Academics were mainly interested in the invasion process, the characteristics associated to invasibility and invasiveness (see Box 1), and in new methods for managing invasions along time and across space (Pooley and Queiroz, 2018).

Though of high importance, approaching alien species and invasions as pure ecological or environmental phenomena, may not be enough to address their complex challenges (Kueffer and Hadorn, 2008; Liu et al., 2007). For instance, in the ecological realm, invaders can impact on ecological attributes and functions that sustain the provision of ecosystem services, while at the same time changing the functioning or quality of other functions and attributes (see Box 3). The benefits or nuisances emerging from these changes in ecosystems can only be recognised in the social realm, specifically trough the variety of values, socio-political conditions, perceptions, attitudes, knowledge and ideas attributed to plant invaders by humans (Essl et al., 2017; Shackleton et al., 2018a; see Box 4). At the same time, the magnitude of species introductions, the type and extent of impacts they have on the ecological system, and the level of people's acceptance of these impacts, depend on ecosystem resilience (and resistance) as well as on human agency and actions (Cote and Nightingale, 2012; Essl et al., 2018; Estevez et al., 2015).

[5] Despite recognition of previous interest on the study of biological invasions (e.g. Vaz et al., 2017a), *The Ecology of Invasions by Animals and Plants* by Charles Elton (1958) is generally considered as the beginning of the systematic scientific study of biological invasions (Richardson and Pysek, 2008).

Box 3: The ecological realm in the ecosystem service cascade

In this thesis, the interactions between ecosystem services and invasions are grounded on the ecosystem services' cascade model by (Haines-Young and Potschin, 2010; Spangenberg et al., 2014; see Figure B3).

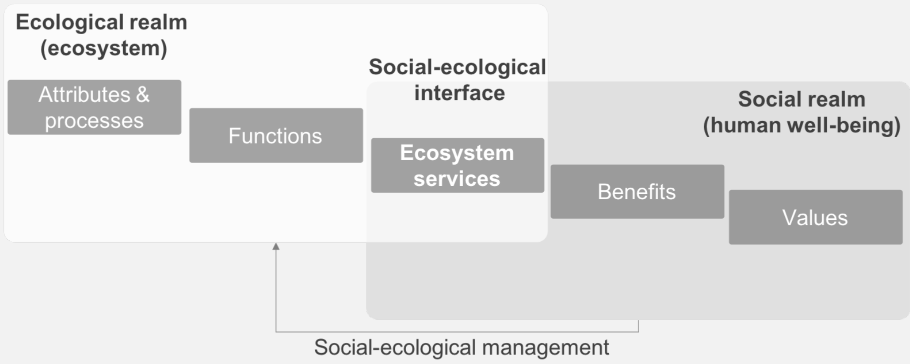

Figure B3. Representation of the ecological realm in a simplified ecosystem services cascade model, showing the relations between ecosystem attributes and processes, ecosystem functions, and the resulting ecosystem services (from Haines-Young and Potschin, 2010).

In this model, the provision of ecosystem services depends on the attributes, processes and functions generated by the ecosystem (i.e. the ecological realm), which can be defined as (Braat, 2014; de Groot et al., 2002):

Attributes: biotic (living organisms) and abiotic (chemical and physical) components of ecosystems, including the characteristics and patterns of the biophysical structure and biodiversity;
Processes: interactions between biotic and abiotic components of ecosystems (i.e. attributes) through the universal driving forces of matter and energy;
Functions: capacity of ecosystem processes and attributes to support goods and services that satisfy human needs, directly or indirectly, including regulation (e.g. bio-geochemical cycles), habitat (e.g. conservation of biological diversity), production (e.g. biomass generated by photosynthesis), and information (e.g. colour and smell) functions.

In this realm, ecosystems may have distinct levels of multifunctionality, depending on biodiversity attributes and processes (Harrison et al., 2014), often delivering bundles of ecosystem services and leading to synergies or trade-offs of benefits (Bennett et al., 2009; Berry et al., 2016; see Box 4):

Bundles: a set of associated ecosystem services that are linked to a given ecosystem and that usually appear together repeatedly in time and/or space;
Synergies: the use of one service increases the benefits supplied by another service through time**;**
Trade-offs: the use of one service decreases the benefits supplied by another service, now or in the future.

In this context, the increasing recognition of biological invasions as inherent socio-ecological phenomena requires insights from multiple disciplines within collaboration networks (Abrahams et al., 2018; Kueffer, 2010; Kueffer and Hadorn, 2008; Matzek et al., 2014; Vaz et al., 2017a). Interdisciplinarity at the social-ecological interface has been advocated in order to better understand the social mechanisms which define the multi-scale drivers, patterns and processes of invasions (i.e. "the human dimension" *sensu* (McNeely, 2001; see section 1.3.2). Concurrently, technological opportunities, such as those provided by remote sensing, are also being increasingly explored to support more effective management options and solutions (Ricciardi et al., 2017; Vaz et al., 2017b; see section 1.3.3).

1.3.2. The expansion of social-ecological perspectives in invasion science

It is now largely accepted that understanding and addressing the human dimension of alien species is paramount to achieve successful knowledge, communication and management towards biological invasions (e.g. Estevez et al., 2015; Frawley and McCalman, 2014; McNeely, 2001; Queiroz and Pooley, 2018; Ricciardi and Ryan, 2018). Despite the importance of social perspectives, the human dimension of invasions only gained significant consideration since the 2000s (Vaz et al., 2017a), namely through an explicit anthropocentric view in the publication of *The Great Reshuffling* (McNeely, 2001). Furthermore, grounded on human valuation, the emergence of the ecosystem services arena provided new research directions for the interdisciplinary study of invasions (Simberloff et al., 2013; van Wilgen et al., 2008).

There are several dimensions in the study of alien species from a social perspective. They include, for instance, historical overviews and metaphors on the relations between human and alien species introductions (Larson, 2005; Rotherham, 2011), philosophical and ethical debates over concepts and ways of thinking in invasion science (Ricciardi and Ryan, 2018), and the recognition of multiple socio-economic drivers of invasions (Kueffer, 2013). Also, social perspectives have been widely adopted to better evaluate the social, cultural and economic impacts of alien species (Essl et al., 2017), and the clarification of human interests, values, perceptions, and attitudes towards these species (Estevez et al., 2015; Kueffer and Kull, 2017; Larson et al., 2011; Shackleton et al., 2018a).

Particularly during the last decade, economists, geographers, historians, philosophers, politicians, and sociologists have been called to focus on how the diverse social actors and factors promote, hinder and shape the invasion process, e.g. through social media, across immigration borders or through trade (Reino et al., 2017; Rodríguez-Labajos et al., 2009). Also, academics have been examining how people perceive alien species, while accounting for cultural influences and normative issues, such as xenophobic and ethical standpoints (Essl et al., 2017; Shackleton et al., 2018a; Tassin and Kull, 2015). This has been targeted by disentangling human believes, attitudes, intentions and behaviours (see Box 4), as well as their cultural and knowledge differences towards invasive species (Buijs et al., 2012; Clavero, 2014; Heger et al., 2013).

By contemplating the social perspective of invasions, academics have contributed to establish more integrative management solutions, by considering conflicts of interest, work capacity, efficiency, and legitimacy of people that manage alien species or invaded areas (Essl et al., 2017; Estevez et al., 2015; Simberloff et al., 2013). Examples include public participation and citizen science methods for the detection, surveillance and monitoring of invasions (Ricciardi et al., 2017), inspection regimes for reducing invasion risk (Ameden et al., 2009), participatory and deliberative processes with key stakeholders (Humair et al., 2014) and approaches focused on public advertising and outreach (Marchante and Marchante, 2016).

Box 4: The social realm in the ecosystem service arena

In the ecosystem service cascade (see Box 3), ecosystem services depend not only on the ecological realm, but also on the capability of people to use, perceive and value ecosystem attributes and functions. The ecological realm is 'value-free', yet it is trough human valuation that ecosystem attributes and functions contribute to or are perceived as promoters of human well-being, at the social realm (see Figure B4).

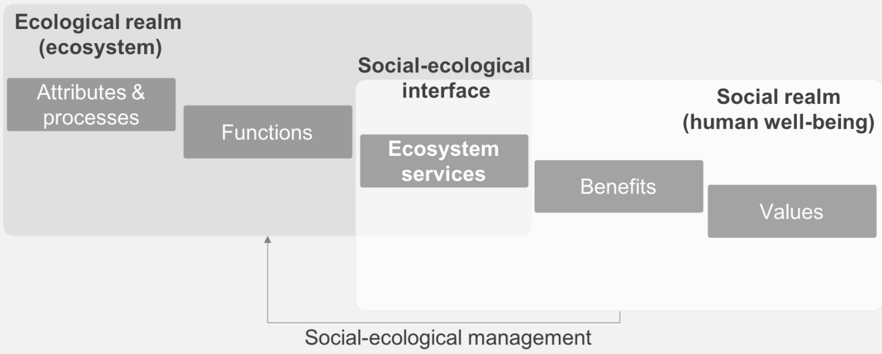

Figure B4. Representation of a simplified ecosystem services' cascade model, showing how the ecological and social realms jointly shape ecosystem services (from Haines-Young and Potschin, 2010).

For instance, alien invaders have the ability to change ecosystem attributes and processes, therefore altering the provision of ecosystem services. Nevertheless, the (lack of) benefits resulting from these changes depend on:

Valuation: as the mental process which includes the assessment of situations and decision-making on whether to act or refrain from action (Braat, 2014); differs from valuing which is the informal, largely implicit process not bound to any particular setting (Kenter et al., 2015);
Values: as the expression or deduction of the valuation process (Kenter et al., 2016), including transcendental (as guiding principles, namely ethics and normative beliefs), contextual (as opinions about worth or importance, which depend on an object of value and its context, as well as individual attitudes and preferences) or value-indicators (as expressions of value in commonly understood units such as money, ratings, indices);
Benefits: realisation and perception of an improved state in the well-being dimension. For instance, air quality regulation is a service, improved health is a benefit and people's perception of the importance of this is a contextual value (though health can be seen as a transcendental value).

Sometimes, it might be difficult to separate services from benefits, particularly when it comes to cultural ecosystem services (see e.g. Díaz et al., 2018; Fish et al., 2016). Nevertheless, the co-production (or joint production) of ecosystem services depends on management actions conducted by humans in the ecological (e.g. removal of plant invaders and landscape restoration) and social (e.g. context-dependent dimensions of economics, politics, technology and culture) realms. Among many others, the social management of ecosystems services, such as those associated to invasions, depends on (Kenter et al., 2015; Shackleton et al., 2018a)):

Perception: the process wherein people select, organise, interpret, retrieve and respond to the information from the world around them;
Attitudes: the favourable or unfavourable evaluations of an object or issue;
Beliefs: propositions that are accepted as true, without value judgement;
Behaviours: actions adopted considering a given value.

1.3.3. The remote sensing revolution in invasion science

Despite the great worth of social perspectives for advancing research on alien and invasive species, management efforts must consider the ecological realm (Ehrenfeld, 2010; Simberloff et al., 2013). This is because the establishment of invasive species in a given area often produces impacts on the attributes, processes and functions of ecosystems (see Box 3), such as biomass production, water regulation and soil formation (Dickie et al., 2014; van Wilgen and Richardson, 2014). These ecological changes are dynamic, complex and sometimes irreversible, depending on the spatial, temporal and social-ecological contexts (Eviner et al., 2012). Therefore, management interventions must consider, among others, the stage of the invasion process (Simberloff, 2015; Simberloff et al., 2013), the characteristics of the invasive species, their invasion potential (invasiveness), the extent and time since invasion, and the features of the invaded environment (invasibility; Gaertner et al., 2014; Jeschke et al., 2014; Vilà et al., 2011).

For long, management efforts (see Box 5) have tried to control or eradicate invasive species and to contain their impacts with difficult, costly and often ineffective management options (Meyerson and Mooney, 2007; Simberloff et al., 2013). Pragmatic management options also aim to prevent invasion risks and early detect new invasions (Juanes, 2018; Vicente et al., 2016), as well as to adapt to their impacts (Kueffer, 2017; Vaz et al., 2017b). This raises the need to devise technological solutions so as to help on the implementation of time- and cost-efficient options for invasion management (Ricciardi et al., 2017). The search for the integration of technological solutions in invasion science has mostly emerged in the 1990s, with a growing interest in predictive models of invasions, such as species distribution models (Buchadas et al., 2017; Guisan and Zimmermann, 2000; Hui and Richardson, 2017). Among others, technological advances have moved towards the fields of biotechnology, gene-editing and eDNA, which, though useful, might constitute risky options (Ricciardi et al., 2017).

Box 5: Managing biological invasions and their impacts

Particularly since the middle of the 19th century, biological invasions became a prominent issue in many scientific and political initiatives. Among others, emblematic initiatives comprise the *International Plant Protection Convention* (IPPC, in 1951) on preventing and controlling the spread of plant pests (many of which are invasive alien species) and the *Convention on Biological Diversity* (CBD, 1992), particularly to "*prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats, or species*" (Article 8h). Following the CBD, the consortium *Global Invasive Species Programme* (GISP), led by the *Scientific Committee on Problems of the Environment* (SCOPE), the *Centre for Agriculture and Bioscience International* (CABI), and the *World Conservation Union* (IUCN), in partnership with the *United Nations Environment Programme* (UNEP), was established to provide scientific support for decision-making on invasive species.

More recently, the 2030 *Agenda for Sustainable Development* of the United Nations, highlights the need "*to prevent the introduction and significantly reduce the impact of invasive alien species on land and water ecosystems and control or eradicate the priority species*" (Target 15.8). The European Union's (EU) regulation for the prevention and management of invasive alien species also constitutes a pioneer initiative in the protection of Europe's natural capital (Regulation EU 1143/2014). This regulation is already transposed to Portugal, which, since the 1990s, recognised alien species as a threat to the services provided by ecosystems (Decree-Law 565/1999). Finally, the recent *Portuguese Strategy for Nature and Biodiversity Conservation* (Decree-Law 31/2018, up to 2030) emphasises the role of alien invasive species on ecosystem services.

There is a diversity of publications on invasion management, and different approaches have been pointed as relevant for dealing with invasive species. For simplicity, management is here considered in its broad sense, as the act of controlling or organising something (Cambridge dictionary). Biosecurity is a related concept, as the management of risks posed by organisms to the economy, environment, and human health through exclusion, mitigation, adaptation, control, and eradication (Pyšek and Richardson, 2010). Throughout this thesis, the most applied concepts are (Funk et al., 2013; Pyšek and Richardson, 2010; Simberloff et al., 2013):

Risk assessment: to identify species invasion potential and how prone are the areas to invasions;
Prevention: to avoid invasions and their impacts;
Early detection (and rapid response): to detect invasive species quickly and respond fast to their presence;
Eradication: removal of the invasive species from a given area;
Control: to contain species populations or their impacts within certain limits;
Mitigation: to reduce the magnitude of invasions and of their impacts;
Restoration: to promote the resilience and recovery of the invaded area;
Adaptation: to adapt to changes derived from invaders without halting their probability of occurrence, while compensating for the damages;
Monitoring: to follow the progress of the invasion process over a period of time.

Furthermore, Ecology is currently witnessing a “remote sensing revolution” (Kwok, 2018), which has proved useful in invasion research, namely in the identification of invasive species and invaded ecosystems (Dvořák et al., 2015; Müllerová et al., 2017; Müllerová et al., 2013), in predicting their potential distribution (He et al., 2015; Rocchini et al., 2015), and in evaluating landscape invasibility (Große-Stoltenberg et al., 2018; Truong et al., 2017).

In fact, with the increasing access to airborne imagery and information from phenological sensors and the development of satellite Earth observation, remote sensing (see Box 6) is becoming key to understand invasion impacts on ecosystem attributes, functions and their related services (Alcaraz-Segura et al., 2013; de Araujo Barbosa et al., 2015; Hellmann et al., 2017; Vaz et al., 2018a). As technology evolves, big data are becoming available for modelling purposes (Rocchini et al., 2015) in emerging open-source and user-friendly platforms with increasing processing capability (e.g. Google Earth Engine). Alongside data from other disciplines, such as ecology, environmental sciences and social sciences (e.g. social media, citizen science; Kissling et al., 2018), remote sensing will strengthen the horizon for invasion science’s interdisciplinarity (Vaz et al., 2018a).

Box 6: Remote sensing: a brief overview of principles, platforms and data

Remote sensing is here understood as the process of capturing information about an object without contacting it directly. A related term often used, i.e. Earth observation, is the gathering of information about the Earth's surface via remote sensing technologies (Kwok, 2018; Murray et al., 2018). Remote sensing technologies include sensors on-board satellites (spaceborne) and aerial platforms such as aircrafts or drones (airborne), as well as on-ground infrastructures (e.g. ground vehicles, towers or buildings) or even humans (e.g. through field spectrometers or digital cameras). When complemented by in-field assessments, remote sensing data can be used to investigate the drivers, patterns, processes and impacts on the Earth's biological, physical or chemical systems (Borre et al., 2011), namely those associated to alien species. This is due to the capacity of remote sensors to detect and measure the energy reflected or emitted from the Earth's surface as radiation in the different domains of the electromagnetic spectrum (Lavender and Lavender, 2016).

There are two general types of sensors that measure the intensity of a signal (i.e. 'band' or 'channel') within the electromagnetic spectrum, differing in the way they operate: active or passive sensors. Active sensors emit a pulse and later measure the energy returned, scattered or bounced back to a detector (e.g. LiDAR technology; Turner et al., 2003), independently from the sun's electromagnetic energy or thermal energy of the Earth (Jensen, 2000). Passive sensors do not emit radiation; instead, they measure the electromagnetic radiation that is reflected from the Sun or is directly emitted by the Earth's surface (Turner et al., 2003). In this process, different properties of remote sensing data can be recognised (Jensen, 2000):

Radiance: the amount of radiation leaving a source (in a given area) that is captured by the sensor;
Reflectance: the ratio between the electromagnetic energy coming from the sun and the electromagnetic energy going back to the sensor;
Scattering: the process through which atmospheric particles interact with and provoke redirections of electromagnetic radiation from the original path;
Absorption: the process in which molecules absorb energy in specific electromagnetic bands, thereby influencing parts of the spectrum available for remote sensing interpretations.

The combination of the previous properties allows to obtain remote sensing imagery useful to: discriminate Earth's attributes (e.g. invaded or invasive vegetation), evaluate ecological functions (e.g. through spectral vegetation signals of stress or functional indices of performance, such as NDVI) or predict environmental processes through models. The quality of remote sensing data depends on (He et al., 2015; Jensen, 2000; Lavender and Lavender, 2016):

Spatial resolution: determines the size of the smallest object that can be recognised in the image;
Spatial coverage: the total area covered by one image, in proportion to the total field-of-view of the sensor;
Spectral resolution: the number of bands in the electromagnetic spectrum that the sensor can measure;
Radiometric resolution: the smallest difference in levels of energy that the sensor can distinguish;
Temporal resolution (or revisit time): the time (hours or days) it takes for a sensor to return to collect data from exactly the same place.

These characteristics, together with data accessibility and users' capability to process and analyse big imagery data, determine the usefulness of the various remote sensing sensors for assessing invasions and how they change the ecological attributes and processes that underlie ecosystem services, at the ecological realm (Alcaraz-Segura et al., 2013; de Araujo Barbosa et al., 2015).

1.4. RESEARCH MOTIVATION AND OBJECTIVES, AND THESIS STRUCTURE

1.4.1. Motivation and objectives

As previously described, the increasing recognition that humanity has entered into a new epoch in which mankind is a key player at a planetary state (i.e., the Anthropocene) highlights the significance of the global biodiversity crisis as well as the magnitude of global anthropogenic changes (Lewis and Maslin, 2015). Among the various fingerprints of this human epoch, the large-scale redistribution of species worldwide (i.e. "the great reshuffling"; McNeely et al., 2001) has promoted a global homogenisation of the Earth's biota (Chew, 2015; Kueffer, 2017).

The global exchange of species is a phenomenon that may result in a modern Pangaea of biodiversity (after Baiser et al., 2012). It's not just the biodiversity patterns that are reshaped by the movement of alien species, but also the structure and functioning of ecosystems, particularly where those species invade and become dominant. This brings consequences for the services (and disservices) that ecosystems provide, and hence nature's benefits (and nuisances) that people use, value and perceive. Concurrently, novel socioeconomic and cultural opportunities, often driven by long-term invasions, trigger important changes in the organization and functioning of social systems translated by distinct interests, attitudes and behaviours towards the (non-)management of alien species (Kull et al., 2018; Shackleton et al., 2018a).

From a social-ecological viewpoint, biological invasions can be seen as a globalised *dance* between aliens and humans. Therefore, acquiring efficiency and practicability in management decisions designated for invasions must be considered alongside the human dimension of alien species (McNeely, 2001). Also, the "information and technological age" reveals powerful information and opportunities to examine the different steps in the alien-human *dance* (Ricciardi et al., 2017). Consequently, a growing appeal for interdisciplinarity has been made on how to deal with invasions, both in academic studies and for real-world solutions (Bennett et al., 2017; Estevez et al., 2015; Vaz et al., 2017a). Within this context, a focus on the dynamics of nature's contributions to human well-being (i.e., ecosystem services and disservices) may provide an arena for interdisciplinary thinking, that moves beyond the classical recognition of invasions as ecological phenomena in order to reframe invasions as a globalised social-ecological phenomenon (Vaz et al., 2017a).

International policy and monitoring initiatives are increasingly recognising invasions as a social-ecological phenomenon with implications for human well-being (see Box 6). Common to these initiatives is the need to produce knowledge and develop management tools that can efficiently tackle invasions and their impacts on ecosystem services. Management options will need to consider an interdisciplinary lens. On the one hand, this lens should allow at understanding people's perceptions, tolerances and preferences towards alien invasive species, so as to deliberate risks and opportunities for human well-being, under minimum management conflicts (Essl et al., 2017; Estevez et al., 2015; Shackleton et al., 2018a; Shackleton et al., 2018b). On the other hand, the lens should be able to monitor invasion patterns and processes through further development of technological approaches (e.g. Earth observations, modelling tools, social media) applicable at multiple spatial and temporal scales, in order to track the ecological changes induced by the occurrence and spread of invasive species (Müllerová et al., 2017; Ricciardi et al., 2017; Vaz et al., 2018b).

This thesis aims to contribute **to improve the understanding and management of biological invasions as a social-ecological phenomenon**, following and combining two major lines of investigation:

Social-ecological approaches: the societal benefits and nuisances related to biological invasions are space and time dependent and are influenced by human perception and management. Therefore, a better understanding and management of biological invasions and their invaded ecosystems is expected to be achieved by combining different disciplinary views and scientific methods, including conservation and environmental sciences, social sciences and humanities, among others.

Remote sensing advances: different missions of Earth observations are providing great amounts of information on the Earth's features and processes, with a range of applications in environmental and social-ecological analysis. Taking advantage from this information is expected to improve our ability to detect and anticipate invasions, and especially to support the assessment of ecological changes induced by invasive species and their related human-invader interactions.

These two lines of investigation were explored in parallel and then combined, following **three main questions**:

- Can an ecosystem (dis)services framework, *grounded on the benefits and nuisances for human well-being*, improve the understanding and management of biological invasions as a social-ecological phenomenon?
- How have invasion research and management taken advantage of the opportunities provided by remote sensing advances, and *how can they further benefit from those opportunities*?
- Can *social-ecological approaches and remote sensing be combined in integrative frameworks* that effectively improve the future assessment and management of invasions?

To target these questions, seven studies (reviews, letters or standard research articles) were developed, each corresponding to a scientific manuscript published or in consideration in peer-reviewed international journals. Common to the seven studies is the application of research synthesis methods (see Box 7). Given their relevance in the Anthropocene, the studies emphasise alien plant invasions (Kueffer, 2017), and particularly alien tree species - which are listed amongst the most challenging plant invasions worldwide (Lowe et al., 2000). A special attention is given to the Iberian Peninsula, and to the only National Park in Portugal, where many alien and invasive tree species have been introduced and promoted. The different chapters are briefly described in the next section (see 1.4.2. Thesis structure).

Box 7. Research synthesis: definition, types and the review process

Research synthesis is the process of synthesising research findings (Haddaway et al., 2015; Lortie, 2014). The term ‘research synthesis’ emerged in the early 1990s, particularly in health sciences (Grant and Booth, 2009) to handle big data and report the outcomes from multiple studies (Lortie, 2014). In two decades, alongside big data production and sharing, the use of research synthesis has expanded to other disciplines such as ecology and social sciences, to summarise the ever-increasing rate of evidence being published, to clarify controversies, to identify general hypotheses, patterns and research gaps, and to communicate findings among different publics and disciplines (Haddaway et al., 2015). There are different types of research synthesis, differing in their goals and methodology (Grant and Booth, 2009); the ones most often applied in social-ecological research include (Doerr et al., 2015; Lortie, 2014):

Critical review: a review that demonstrates an extensively knowledge of literature and critically evaluates its content and/or quality to include conceptual innovation beyond mere descriptions;
Systematic review: a way to systematically search, appraise and synthesis research evidence, often adopting standard and objective protocols on how to conduct a review;
Meta-analysis: a technique that statistically combines the results of quantitative studies to provide a more precise effect of the outcomes from the review.

Different methods can be applied in the review of scientific evidence. Depending on the research question underlying the review, a combination of methods may be used for exploring quantitative or qualitative information from published evidence (Grant and Booth, 2009). Nevertheless, the process of research synthesis usually includes the following general steps (Doerr et al., 2015; Grant and Booth, 2009):

Preparing the review: establishing the question, gathering a team of experts, developing and testing a search strategy, and describing the review plan, including the search strategy, data collection and analysis;
Searching for studies: conducting the search of relevant studies (e.g. online literature using Scopus), filter the studies through inclusion and exclusion criteria**,** and evaluate the quality of the obtained studies;
Reviewing the information: obtaining information from the studies needed to answer the research question and analyse the obtained information; this analysis may be qualitative (e.g. critical or narrative analysis) or quantitative (counting, statistical tests or meta-analysis).

1.4.2. Thesis structure

This thesis includes eight chapters, starting with a general introduction (**chapter 1**) that outlines the context, the general approach, the motivation and the objectives of this thesis.

Then, **chapter 2** describes a new conceptual framework to integrate ecosystem services and disservices for human well-being. The framework is illustrated considering the multiple benefits and nuisances provided by plant invasions worldwide. It also accounts for the role of social-ecological management in the valuation of ecosystem services and disservices associated to plant invasions. Grounded in this framework, chapter 2 further proposes a precautionary management hierarchy to be used when considering ecosystem services and disservices promoted by invasions.

To advance the state-of-knowledge in invasion research and explore potential management applications, two complementary approaches are then followed, grounded on social-ecological perspectives (chapters 3 & 4) and on remote sensing advances (chapters 5 & 6).

Interdisciplinary approaches to the study of invasions as a social-ecological phenomenon are reviewed in **chapter 3**. The chapter provides an overview on how to foster collaboration and insights from social sciences to understand the role of biological invasions as promoters or disrupters of ecosystem services, supporting management actions that meet with human well-being. To do so, the chapter starts by reviewing the extent to which interdisciplinarity has featured in invasion science over the last half-century, focusing on the integration of ecological and social sciences. Based on this quantitative review and temporal narrative, the chapter proposes pathways for promoting progress in invasion research and management through an explicit social-ecological way.

A possible way to address the social-ecological side of invasions is to focus on the study of cultural ecosystem services and how they may be affected by invasions. In this context, **chapter 4** adopts a social-ecological perspective to assess the effects of alien trees – many of which are among the most challenging invasive species – on multiple cultural services. The study includes the use of meta-analysis statistics to evaluate photographic, internet and catalogue data at the regional level in the Iberian Peninsula. Then, it evaluates whether regional variations of alien tree effects differ between countries (i.e., Portugal and Spain) and along different contexts related to land cover and management, socio-economy, human well-being, and climate. By doing so, the chapter ends by discussing perspectives for informed management of alien trees in the Iberian Peninsula.

Chapters 5 and 6 focus on the opportunities offered by remote sensing to assess aliens from space, namely regarding their detection and prediction as well as the assessment of their impacts on the functions and services of invaded ecosystems or landscapes.

First, **chapter 5** reviews and discusses the progress, current state and opportunities of remote sensing applications in the research and management of plant invasions. The rationale underlying this review is grounded on the management framework described in chapter 2 and covers the contributions of remote sensing for supporting the management of potential impacts of plant invaders on ecosystem services and disservices. Finally, the chapter discusses possible ways of taking advantage of current and future Earth observation methods and missions to improve invasion science.

Then, **chapter 6** explores the potential of remote sensing products to evaluate the seasonal contribution of alien tree species to cultural ecosystem services. This evaluation takes place at the only National Park in the country ("Peneda-Gerês"), for which management efforts need to consider both natural and cultural assets. Spatial and temporal (i.e. seasonal) differences in the contribution of alien trees are evaluated considering the environmental context (associated to accessibility and wilderness) and landscape visual-sensory features (related to spatial diversity, colour and ecosystem functioning). The chapter concludes with a series of recommendations for informed management regarding alien trees, focused on the safeguard of natural capital along with recreational benefits.

Building on the previous social-ecological and remote sensing perspectives, **chapter 7** presents potential ways forward in the research of ecosystem services and in the science of biological invasions. This chapter includes two sections, both following a letter type of narrative. The first section (7.1) discusses the opportunities that remote sensing offers for the understanding of cultural ecosystem services at a planetary scale. The second section (7.2) argues the consideration of remote sensing as an emerging issue of increasing relevance and applicability in invasion science.

Finally, **chapter 8** provides an integrative discussion of the previous chapters. A particular emphasis is given to the usefulness of advancing interdisciplinarity approaches for the study and management of invasive species. The implications of bringing together social-ecological perspectives and remote sensing opportunities for managing invasions with a focus on ecosystem services and disservices are also emphasised. The chapter ends with general conclusions and some final remarks. Figure 1.1 illustrates the sequence and the links between the several chapters of this thesis.

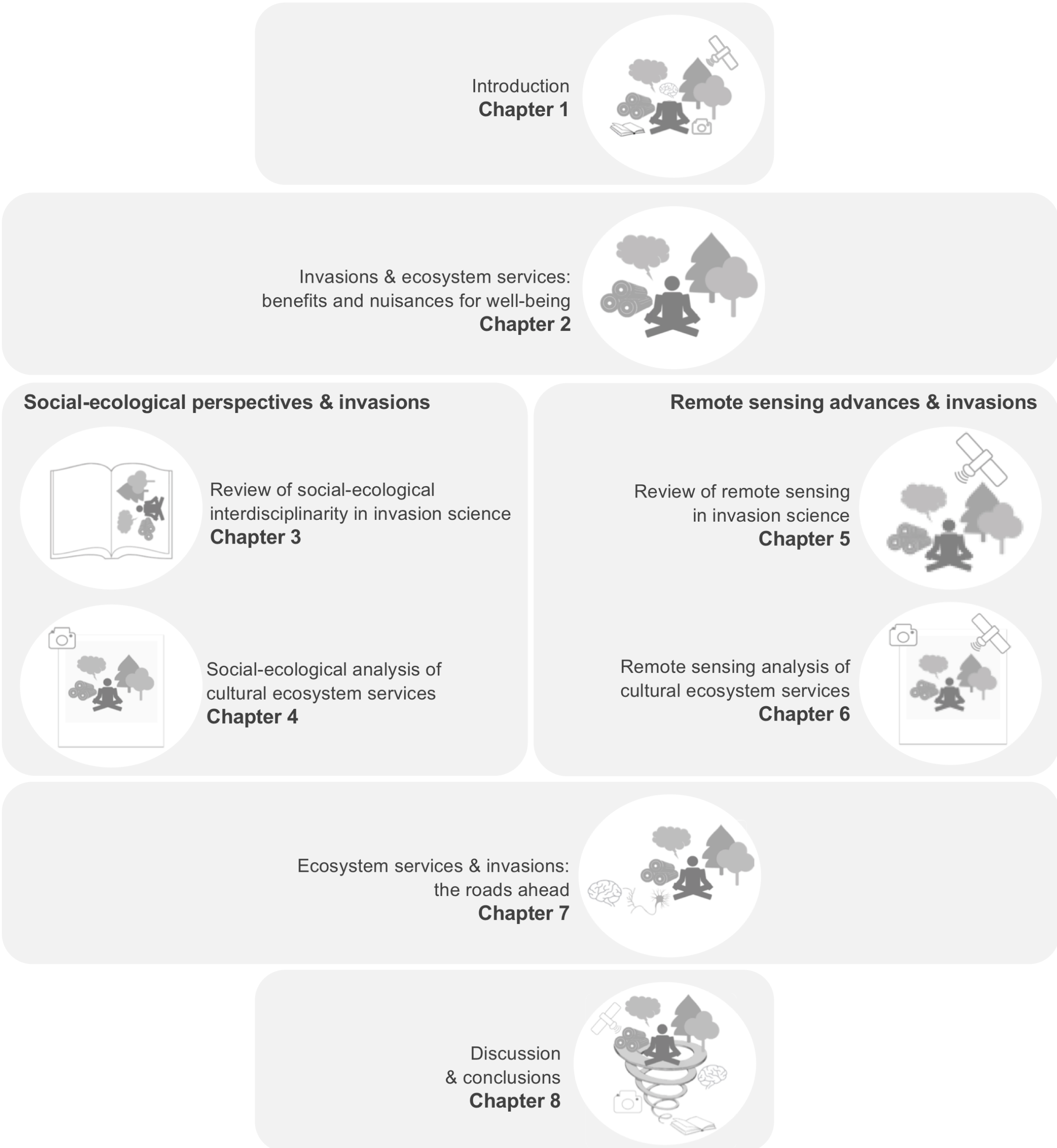

Figure 1.1. Representation of the thesis structure with eight chapters aiming to understand the role of biological invasions as drivers of human well-being (chapter 2), considering social-ecological perspectives (chapters 3 & 4) and remote sensing opportunities (chapters 5 & 6). Possible pathways for future research on ecosystem services and invasions are discussed in chapter 7. A general discussion and the main conclusions are presented in chapter 8.

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CHAPTER 2. INVASIONS & ECOSYSTEM SERVICES: BENEFITS AND NUISANCES FOR HUMAN WELL-BEING

DISCLAIMER

This chapter is an original contribution of this thesis, published as a review paper in journal *Ecosystem Services* under the title "*Integrating ecosystem services and disservices: insights from plant invasions*". The paper was considered the "Best Sustainable Development Article 2017", by the International Sustainable Development Research Society (ISDRS). The paper had the contribution of the following authors:

Ana Sofia Vaz[a,b], Christoph Kueffer[c,d], Christian A. Kull[e,f], David M. Richardson[d], Joana R. Vicente[a,g], Ingolf Kühn[h,i,j], Matthias Schröter[j,k], Jennifer Hauck[l,m], Aletta Bonn[j,k,n], João P. Honrado[a,b]

[a]Research Network in Biodiversity and Evolutionary Biology, Research Centre in Biodiversity and Genetic Resources (InBIO-CIBIO), Campus Agrário de Vairão, Rua Padre Armando Quintas, PT4485-661 Vairão, Portugal;

[b]Faculty of Sciences, University of Porto, Rua do Campo Alegre, s/n, PT4169-007 Porto, Portugal;

[c]Institute of Integrative Biology, ETH Zurich, Universitätstrasse 16, CH-8092 Zurich, Switzerland;

[d]Centre for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, Matieland 7602, South Africa;

[e]Institute of Geography and Sustainability, University of Lausanne, 1015 Lausanne, Switzerland;

[f]Centre for Geography and Environmental Science, Monash University, Melbourne, Vic 3800, Australia;

[g]Laboratory of Applied Ecology, CITAB - Centre for the Research and Technology of Agro-Environment and Biological Sciences, University of Trás-os- Montes e Alto Douro, Vila Real, Portugal;

[h]Department of Community Ecology, UFZ: Helmholtz Centre for Environmental Research, Theodor-Lieser-Str. 4, Halle, Germany;

[i]Geobotany and Botanical Garden, Martin-Luther-University Halle-Wittenberg, Am Kirchtor 1, 06108 Halle, Germany;

[j]German Centre for Integrative Biodiversity Research (iDiv) Halle-Jena-Leipzig, Deutscher Platz 5e, 04103 Leipzig, Germany;

[k]Department of Ecosystem Services, UFZ - Helmholtz Centre for Environmental Research, Permoserstr. 15, 04318 Leipzig, Germany;

[l]UFZ - Helmholtz Centre for Environmental Research, Permoserstr. 15, 04318 Leipzig, Germany;

[m]CoKnow Consulting, Mühlweg 3, 04838 Jesewitz, Germany;

[n]Institute of Ecology, Friedrich Schiller University Jena, Dornburger Str. 159, 07743 Jena, Germany.

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ABSTRACT

There is growing interest in ecosystem disservices, i.e. the negative effects of ecosystems on humans. The focus on disservices has been controversial because of the lack of clarity on how to disentangle ecosystem services and disservices related to human well-being. A perspective that considers both services and disservices is needed to inform objective decision-making. We propose a comprehensive typology of ecosystem disservices and present a framework for integrating ecosystem services and disservices for human well-being linked to ecosystem functioning. Our treatment is underpinned by three key assumptions: (1) ecosystem attributes and functions are value-free; (2) the perception of benefits or nuisances are however dependent on societal context, and preferences and actions by societal actors may trigger, enhance or alleviate benefits or nuisances derived from ecosystems; and (3) the notion of disservices must account for the role of human management in assessments of ecosystem values, i.e. the social and technological measures that identify, protect, promote or restore desirable levels of services, and concurrently minimise, mitigate or adapt to disservices. We illustrate our ideas with examples from plant invasions as a complex social-ecological phenomenon.

Keywords: Biological invasions; Ecosystem function; Human valuation; Invasive species; Social-ecological management

2.1. INTRODUCTION

The concept of ecosystem services has emerged from the recognition that complex interactions in ecosystems can result in flows of energy, matter and information, which contribute to human well-being. Examples include fostering basic needs through food, fibre and energy provision as well as regulation services (e.g. carbon sequestration, pollination, pest control) and contributions to cultural aspects of well-being (Agarwala et al., 2014; Díaz et al., 2015; MA, 2005; Smith et al., 2013). The focus on ecosystem services has created an additional perspective which differs from, and is complementary to, traditional conservation policies for ensuring the sustainable use and the protection of ecosystems (Agarwala et al., 2014; Bonn et al., 2016; Brown and Westaway, 2011). Yet, one of the major recurring points of criticism of the notion of ecosystem service is that it often considers only the beneficial outputs of ecosystems and ignores unpleasant, unwanted or economically harmful effects (Lyytimäki and Sipilä, 2009; Lyytimäki, 2014; Schröter et al., 2014). These negative sides of ecosystems have been termed ecosystem disservices. Following Shackleton et al. (2016: p. 590), ecosystem disservices are "*the ecosystem generated functions, processes and attributes that result in perceived or actual negative impacts on human well-being*".

Ecosystem disservices can be produced, for example, by biological invasions (Shackleton et al., 2016), and by other ecosystem attributes that are perceived as unwanted (Escobedo et al., 2011; Lyytimäki et al., 2008). They are produced by ecosystem functions, such as wildfires or floods, which pose danger to people and, although they may constitute natural processes, can be mitigated or exacerbated through management (Lyytimäki, 2014). The same ecosystem function may be perceived as a service by some people and as a disservice by other people (cf. Saunders and Luck, 2016), depending on, among other things, acquired knowledge, people's behaviours, and overall political, economic and social settings (Rasmussen et al., 2016; Shackleton et al., 2016; Stoll et al., 2015). Configuration of anthropogenic pressures as well as provision and perceptions of ecosystem services and disservices may vary spatially, temporally and between individuals or societal groups (Chan et al., 2012; Shackleton et al., 2016).

The notion of ecosystem disservices has its main roots in urban ecosystem research (Dobbs et al., 2014; Escobedo et al., 2011; Lyytimäki, 2014; Lyytimäki and Sipilä, 2009), particularly in work associated with complex human-environment systems that characterise large cities (von Döhren and Haase, 2015). Ecosystem disservices have been used to evaluate the value of green space for urban residents (Lyytimäki and Sipilä, 2009; Lyytimäki et al., 2008) given that urban green spaces can provide many services but also a range of disservices, from allergenic substances and volatile compounds emitted by vegetation (Dobbs et al., 2014), to blocking of sunlight by trees (Roy et al., 2012), and the presence of wild animals in people's backyards (Lyytimäki, 2014). The notion of ecosystem disservices has also been extended to agricultural systems (e.g. Ma et al., 2015; Schäckermann et al., 2015) to account for problematic aspects of human managed ecosystems (Ma et al., 2015), to denote increases in production costs, e.g. for pest control (Schäckermann et al., 2015; Swinton et al., 2007; Zhang et al., 2007), or other ecological costs arising from animal activities (Kronenberg, 2014; Whelan et al., 2015).

The usefulness of ecosystem disservices has more recently been discussed for other contexts, namely fisheries and forests (see Shackleton et al., 2016). Yet, disservices have seldom been considered in the context of broader social-ecological challenges (Saunders and Luck, 2016; Shackleton et al., 2016), such as plant invasions. Plant invaders provide both benefits (Tassin and Kull, 2015) and nuisances (Simberloff et al., 2013) for human well-being, depending on people's preferences and the spatio-temporal context (Kueffer and Kull, 2017). In some contexts, invasive plants contribute to people's livelihoods, by supporting daily basic needs and economic incomes (Kull et al., 2011), or by enhancing regulating functions, including coastal sediment dynamics and soil protection. In other contexts, however, plant invasions can lead to undesirable outcomes for human well-being. Examples include health problems associated to allergenic compounds or skin irritations, wildfires in non-fire prone areas, or competition with another service-providing species (Fenesi et al., 2015; Gaertner et al., 2014). The beneficial or detrimental impacts of plant invasions can be exacerbated by the magnitude, rate and scale of the invasion process (e.g. Eviner et al., 2012). The same species can promote ecosystem services at some spatio-temporal extent, e.g. trees confined to private gardens, or contribute to disservices' provision at later stages, e.g. trees that become widespread in the wild (the "transient disservices"; Saunders and Luck, 2016). This inevitably depends on (the lack of) human management (Brundu and Richardson, 2016).

Previous attempts to categorise ecosystem disservices have relied on comparisons with pre-established classifications of ecosystem services. For instance, Ma et al. (2014) introduced the term 'provisioning and regulating disservices' to account for soil loss in agricultural systems. Price (2014) used 'supporting and regulating disservices' in the context of forestry. Other authors, mostly in reference to urban ecosystems (von Döhren and Haase, 2015), proposed mixed typologies, based on both the origin and consequences of ecosystem disservices. Escobedo et al. (2011) classified ecosystem disservices as financial (economic costs triggered by ecosystems), social (impacts on human health and fear) or environmental (affecting intrinsic ecosystem attributes). Lyytimäki and Sipila (2009) categorised disservices based on their origin (as social, social-ecological or ecological), and based on the impacted societal actors (individuals, communities, or humankind). More recently, Lyytimäki (2014) categorised disservices with respect to weather-related events and ecosystem functions causing harm, as well as human fears and risks, activities, or aesthetic issues. Despite their usefulness in specific cases, the above-mentioned typologies do not yet provide the means for distinguishing between the occurrence of a perceived negative service, i.e. an ecosystem disservice, and the reduction of an ecosystem services. For instance, a lack of an explicit differentiation of reduced services and genuine disservices led to ambiguity in the literature (Shackleton et al., 2016), e.g. by denoting habitat loss (Zhang et al., 2007) and pesticide output in agricultural systems (Swinton et al., 2007) as ecosystem disservices. To tackle the conceptual problem that reduced ecosystem services are not necessarily ecosystem disservices, Shackleton et al. (2016) classified disservices based on their effects on the economy, physical and mental health, or aesthetic and cultural issues of human well-being.

Although ecosystem disservices have been accounted in the scientific literature (Shapiro and Báldi, 2014), a comprehensive conceptual framework that incorporates both services and disservices is lacking (compare Saunders and Luck, 2016; von Döhren and Haase, 2015). In our view, this framework should address three conceptual issues: (1) nuisances from ecosystems to well-being can either be expressed as reduced services (e.g. decrease of water provision, or reduction of soil erosion protection), or as genuine disservices (e.g., wildfires and pests; see Saunders and Luck, 2016; Shackleton et al., 2016 for reviews); (2) benefits and nuisances should account for human activities, since feedbacks between ecological changes and societal responses may trigger, enhance or reduce either services or disservices; and, (3) an ecosystem disservice framework should facilitate deliberation about both positive and detrimental aspects of ecosystems for human well-being, acknowledging that there is not only one state in nature that can or should be maintained (or restored) through management. Some experts might consider that this likely opens a Pandora's Box (Shackleton et al., 2016). For example, conservationists who place an emphasis on native, wild nature may feel threatened by a concept and associated conceptual model that might be used to justify interventions in landscapes that they value for their lack of anthropogenic imprint (following Kronenberg, 2014; Villa et al., 2014). Yet clearly, explicit negotiations of management priorities might increasingly become unavoidable in coupled social-ecological landscapes. In these negotiations, ecosystem disservices' recognition might contribute to informed ecosystem management approaches and possibly optimised investments to increase both biodiversity and human well-being (Saunders and Luck, 2016; Shackleton et al., 2016; Stoll et al., 2015).

This paper proposes a general conceptual framework of ecosystem disservices. The framework encompasses a detailed typology of different disservices, and it proposes a way to explicitly account for the role of social-ecological management in the valuation of ecosystem services and disservices. To this end, we highlight the importance of acknowledging the interconnected human-ecological nature of ecosystems. We propose to refine a precautionary approach to ecosystem management through a hierarchy: first identify potential ecosystem services and disservices, then protect ecosystem services and avoid or minimise disservices, restore and rehabilitate ecosystem services, and lastly mitigate and adapt to ecosystem disservices. We illustrate our framework with plant invasions as a test case. Finally, we synthesise the wider usefulness of our typology and framework for the future study and management of benefits and nuisances arising from ecosystems.

2.2. METHODS

2.2.1. The ecosystem disservices' typology

To build the ecosystem disservices' typology, a literature search was performed in ISI Web of Science, between May and July 2015 (updated in February 2016). The search string was TOPIC = ("ecosystem* disservice*" OR "environment* disservice*" OR "landscape disservice*" OR "ecologic* disservice*" OR "ecosystem* dis-service*" OR "environment* dis-service*" OR "landscape dis-service*" OR "ecologic* dis-service*"). The time span of our search was 1900-2015. Following recommendations for increasing the reliability of literature reviews (Higgins and Green, 2011), our search was further extended to the first 50 records retrieved by a search on Google Scholar in February 2016. The records retrieved by ISI (number of records, n = 40) and additional records retrieved from Google Scholar (n = 50) were scrutinised and irrelevant records were discarded, e.g. those which only mentioned the word "ecosystem disservice" but did not address their actual assessment or categorisation, or those which simply mentioned ecosystem disservices, but focused on services. We then reviewed the categories presented by each record from the final set of selected publications and organised the examples and categories to produce a common ecosystem disservices' typology. Since our goal was not to conduct an exhaustive literature search on the ecosystem disservice concept, the records indicated in this manuscript are purely illustrative of each disservice category.

We outline our proposed disservice categories in section 2.3. They are grounded on the same premises that underlie ecosystem services, i.e. they influence different dimensions of human well-being (Agarwala et al., 2014; MA, 2005). Since the definition of human well-being is still widely debated, here we consider human well-being as the desirable conditions for an individual or societal group (Jax and Heink, 2015), which depend on: objective attributes related to people's material and social contexts, subjective thoughts, feelings and satisfactions towards life, and psychological responses associated with social connectedness, security, and life satisfaction (Agarwala et al., 2014; Smith et al., 2013). Following Smith et al. (2013) we thus consider the following well-being dimensions: health, including life expectancy and mortality, and physical and mental health conditions; social cohesion, considering physical and emotional links that connect humans in society: education, resulting knowledge and skills; safety and security, as physical, personal and national freedom from harm and financial destabilisation; living standards, as the access to goods, services and resources; leisure time, as pleasurable activities away from work and responsibilities; spiritual and cultural fulfilment, as opportunities to fulfil spiritual and cultural needs; and connection to nature, as personal connectedness to ecosystems and biota. These dimensions contribute to general life satisfaction and happiness (Smith et al., 2013).

2.2.2. The ecosystem services and disservices' framework and categories

The integration of disservices into a general ecosystem service framework, presented in section 2.4, was grounded on the main ideas underlying the 'ecosystem service cascade model' (Haines-Young and Potschin, 2010). This model describes how the biophysical structure of ecosystems sustains the ecological functions and processes needed to provide ecosystem services. These ecosystem services then contribute to the benefits for human well-being with a respective value (see Haines-Young and Potschin, 2010; Spangenberg et al., 2014a for details).

Several initiatives have focused on the assessment or categorisation of ecosystem services. Prominent examples are the *Millennium Ecosystem Assessment* (MA, 2005), *The Economics of Ecosystems and Biodiversity* (TEEB, 2010), the *Mapping and Assessment of Ecosystems and their Services* (MAES, 2013), the *Common International Classification of Ecosystem*

Services (CICES; Haines-Young and Potschin, 2013), and the *Intergovernmental Platform on Biodiversity and Ecosystem Services* (IPBES; Díaz et al., 2015). The ecosystem services' categories adopted in our framework were based on CICES; these are considered applicable to different spatial and thematic scales and are thus context-independent, allowing multi-study comparisons (Haines-Young and Potschin, 2013). CICES provides a five-level, hierarchical typology, the first level of which separates ecosystem services into provisioning, regulating and maintenance, and cultural services (Haines-Young and Potschin, 2013). A comparison of CICES, MA and TEEB classifications is presented in Table S2.1 (Supplementary material I).

2.2.3. The ecosystem disservices typology and the ecosystem service-disservice framework illustrated with plant invasions

A similar procedure to that used in section 2.2.1 was considered to illustrate the ecosystem disservices' typology in line with an ecosystem services' framework for (alien) plant invasions in section 2.5. In this case, the search string was: TOPIC = ("plant invader*" OR "exotic plant*" OR "alien plant*" OR "allochthonous plant" OR "plant invasion*" OR "tree invader*" OR "exotic tree*" OR "alien tree*" OR "tree invasion*") AND ("ecosystem disservice*" OR "environment* disservice*" OR "landscape disservice*" OR "ecologic* disservice*" OR "ecosystem dis-service*" OR "environment* dis-service*" OR "landscape dis-service*" OR "ecologic* dis-service*" OR "ecosystem service*" OR "environment* service*" OR "landscape service*" OR "ecologic* service*"). The records retrieved in ISI (n = 184) were checked for relevance (e.g. excluding topics such as invaders from outer space). Each record was reviewed, and we selected representative records to extract illustrative examples of the effects of plant invasions on human well-being.

2.3. A TYPOLOGY FOR ECOSYSTEM DISSERVICES

Here we propose a detailed typology for ecosystem disservices, considering a wide number of human well-being dimensions which can be negatively impacted by ecosystems in a direct way (Table 2.1). Our typology is based on an expanded definition of ecosystem disservices that considers the direct "*perceived or actual negative impacts on human well-being*" (after Shackleton et al., 2016).

Table 2.1. The proposed typology of ecosystem disservices with examples from the literature.

Ecosystem disservices	Key references
***Health disservices**: affecting human health*	
- Pollen release that provokes allergic reactions or intoxications; - Animal bites (with or without poison) on humans; - Zoonotic diseases transmitted to humans; - Direct attacks by wild animals causing human injury or death; - Plants that cause irritation when consumed by humans; - Bacteria and virus that resist to human antibiotics; - Methane emissions by plants breathed by humans; - Toxins released by algal blooms and consumed by humans.	Baró et al., 2014; Bennett et al., 2010; Dobbs et al., 2014; Escobedo et al., 2011; Gómez-Baggethun and Barton, 2013; Limburg et al., 2010; Lyytimäki, 2014; Lyytimäki and Sipilä, 2009; Lyytimäki et al., 2008; Ma et al., 2015; Roy et al., 2012; Shackleton et al., 2016; Zhang et al., 2007.
***Material disservices**: damaging built infrastructures*	
- Excrement from animals damaging buildings; - Roots of plants damaging streets or pavements; - Leaf litter considered a nuisance, e.g. stains resulting from leaf tannins; - Natural disasters damaging infrastructures*.	Agbenyega et al., 2009; Dobbs et al., 2014; Escobedo et al., 2011; Gómez-Baggethun and Barton, 2013; Limburg et al., 2010; Lyytimäki and Sipilä, 2009; Lyytimäki et al., 2008; Roy et al., 2012; Sagie et al., 2013; Shackleton et al., 2016.
***Security and safety disservices**: disrupting physical, personal, national and financial stabilisation*	
- Fear and risk of getting lost in the wild due to lack of light (e.g. in dense forests); - Fear and risk of attacks by wild animals (e.g. snakes, bears); - Tree branches falling in roads and causing accidents or traffic delays; - Dense vegetation provoking bad visibility in traffic and communication blockage; - Fire-prone vegetation (e.g. dense biomass stands) in otherwise non-fire prone landscapes; - Weather phenomena impacting human life (e.g. loss of life)*; - Wild animals within private facilities (lizards or poisonous spiders inside houses, or crocodile in backyards).	Agbenyega et al., 2009; Bennett et al., 2010; Escobedo et al., 2011; Escobedo et al., 2011; Gómez-Baggethun and Barton, 2013; Limburg et al., 2010; Lyytimäki, 2014; Lyytimäki and Sipilä, 2009; Lyytimäki et al., 2008; Ma et al., 2015; Roy et al., 2012; Shackleton et al., 2016; Zhang et al., 2007.
***Cultural and aesthetic disservices**: impacts on mental/cultural interactions with nature*	
- Species perceived as disgusting and irritating by people; - Species and landscapes considered unpleasant by people; - Unpopular species due to religion, tradition or cultural legacies (e.g. snakes or goats associated with evil); - Emergence of landscape new views by vegetation perceived as unpleasant.	Ango et al., 2014; Escobedo et al., 2011; Gómez-Baggethun and Barton, 2013; Lyytimäki, 2014; Lyytimäki and Sipilä, 2009; Lyytimäki et al., 2008; Roy et al., 2012; Shackleton et al., 2016.
***Leisure and recreation**: causing inhibition of physical interactions with nature*	
- Sounds and smells produced by animals disrupting physical connection with nature;	Escobedo et al., 2011; Gómez-Baggethun and Barton, 2013;

- Presence of weeds, pests or mosquitoes considered unpleasant for recreation; - Blocking of sunlight by vegetation, creating too much shading for leisure activities; - Algal blooms spoiling water courses for sport fishing or water sports; - Habitats associated with the unknown, remoteness or wilderness considered unpleasant for outdoor activities; - Preference for indoor activities due to unsuited surrounding landscapes.	Lyytimäki, 2014; Lyytimäki and Sipilä, 2009; Lyytimäki et al., 2008; Roy et al., 2012; Sagie et al., 2013; Shackleton et al., 2016.

*The incorporation of natural disasters and weather-related events as disservices is still under debate. We followed Shackleton et al. (2016: p. 592), considering that "*hazard events or phenomena that have a link to biological process qualify to be ecosystem disservices*".

The typology includes five categories. The first category comprises *health disservices* and includes the direct consequences resulting from unwanted effects of biota on human health, including the outputs from their existence, e.g. air pollution caused by plant metabolism, viruses and pollen transmission. The second category comprises impacts on physical material for human life (*material disservices*), disrupting social cohesion, and living standards of human well-being. It includes those circumstances in which the physical expansion or introduction of living organisms results in outcomes that contribute directly to degradation of human materials and structures, such as buildings and houses, including traffic and communication infrastructure such as roads, e.g. through vegetation growth or animal excrements. The third category (*security and safety disservices*) considers impacts on the physical, personal and national security and safety of people. It includes all circumstances in which human freedom becomes affected, either through fear e.g. from animal attacks in wild or remote areas, densely vegetated areas such as parks or forests, or due to a perceived higher risk of becoming a victim of a crime; or physical harm that can be enhanced or mitigated by human activities together with natural processes, e.g. caused by falling tree branches or fire. *Cultural and aesthetic disservices* (the fourth category) refer to biota or ecological outcomes that mostly impact on mental enjoyment of/connection to nature: human perception, aesthetics, spiritual, symbolic, cultural and religious values, such as species or landscapes considered as unpleasant. *Leisure and recreation disservices*, the fifth category, relate to ecosystem outputs that inhibit (the willingness for) physical connection with nature, through leisure and recreation activities, for instance by vegetation occurrence obstructing water courses for water sports and other recreational activities in the wild.

The assumption that ecosystem disservices have direct consequences for human well-being allows us to distinguish disservices from situations in which an ecosystem nuisance is instead derived from the reduction of a service (i.e. reduced ecosystem service). For instance, unwanted ecosystems functions can impact on well-recognised provisioning services from agricultural (e.g. pests and weeds affecting crop growth; fungus degrading processed food; Lyytimäki et al., 2008; Schäckermann et al., 2015), forest (e.g. timber quality damaged by fungus; wood damaged by deer rub; Ango et al., 2014; Lyytimäki et al., 2008) or grazing systems (e.g. cattle diseases or poisoning by the consumption of toxic plants; Shackleton et al., 2016). Thus, the impact on people's living standards or the decrease of financial income that emerges from such impacts is determined by reduced ecosystem services, and not necessarily by genuine ecosystem disservices (see section 2.4 for details).

Additionally, our typology also overcomes the attribution of an a-priori normative judgement to ecosystem properties, since the same property can be considered in multiple ecosystem disservice categories. Instead, the framework allows to account for how ecosystem functions (resulting from such properties) impact on well-being. For instance, snakes (as an ecosystem property) can bite (*Health disservices*), degrade infrastructures through excrements (*Material disservices*), give a sense of fear (*Security and safety disservices*), be considered as ugly (*Cultural and aesthetic disservices*), and occupy wild areas used for outdoor leisure (*Leisure and recreation disservices*). At the same time, snakes can contribute to ecosystem services such as pest regulation, source of poison useful for medicinal purposes, and promote physical and intellectual experiences. These examples show the need for an integrative framing of both services and disservices.

2.4. Integrating ecosystem disservices and services into a general framework

2.4.1. The ecosystem services and disservices' framework

We propose a general framework that includes three main components of a social-ecological system to consider both ecosystem services and disservices: the ecological realm, the social realm, and the social-ecological interface (Figure 2.1). Drawing into the flows underlying the ecosystem service cascade model (Haines-Young and Potschin, 2010; Spangenberg et al., 2014a), we argue that the provision of ecosystem services and disservices at the social-ecological interface depends on the attributes and functions generated in the ecological realm, while it contributes to benefits, i.e. increase of human well-being, or nuisances, i.e. reduction of human well-being, in the social realm (Haines-Young and Potschin, 2013; Spangenberg et al., 2014a, 2014b).

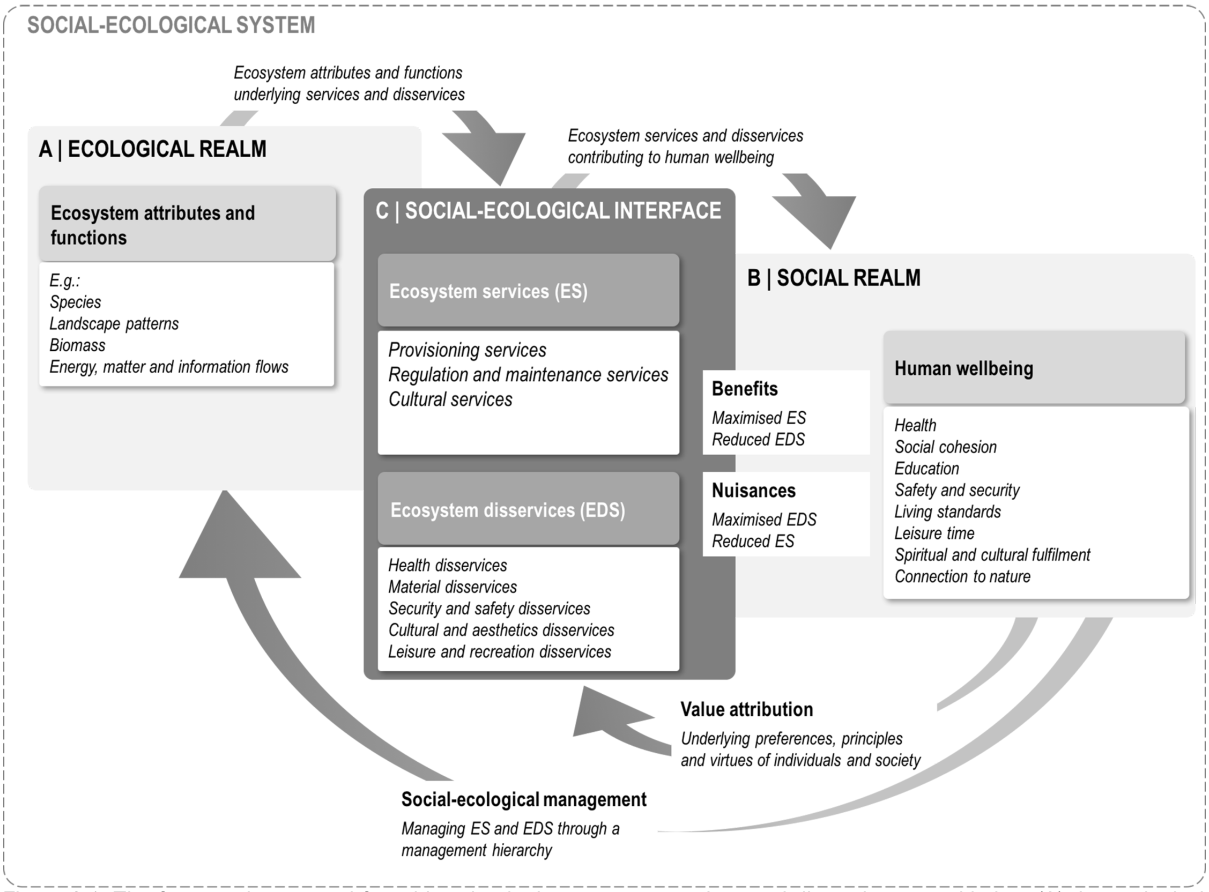

Figure 2.1. The framework proposed for addressing both ecosystem services and disservices, considering: (A) the ecological realm; (B) the social realm; and (C) the social-ecological interface. The framework assumes that the attribution of ecosystem services-disservices depends also on value attribution and social-ecological management.

The perception of benefits and nuisances in the social realm depends on the attribution of human values to the outputs of the ecological realm. However, in extending previous frameworks, we assume that the outputs from the ecological realm are also determined by social-ecological management actions that are interrelated with the valuation resulting from the social realm. To account for the dynamic role of humans in the interdependence of ecological and social processes we introduce the social-ecological interface where the attribution of ecosystem services and disservices happens in specific social-ecological, temporal and spatial contexts.

2.4.2. The ecological realm

The ecological realm (Figure 2.1-A) reflects the set of ecosystem attributes and functions that constitute or generate ecosystem services and disservices (Spangenberg et al., 2014a). It also considers the influence of abiotic components of the ecosystem such as weather events regulated by vegetation (Lyytimäki, 2014; Sagie et al., 2013), in so far as they are situated within the boundaries of an ecosystem (Shackleton et al., 2016). The ecosystem considered can be either a natural or anthropogenically influenced one, such as agricultural or afforested land.

Ecosystem attributes and functions are neither positive nor negative: the same function or attribute can generate, maximise or reduce ecosystem services and disservices (cf. Saunders and Luck, 2016). For instance, the processes underlying a tree's carbon cycle can contribute to climate regulation through carbon sequestration (e.g. Baró et al., 2014), or produce volatile organic compounds, contributing to air pollution and human health problems (i.e. *Health disservices*; e.g. Roy et al., 2012). This depends on the multi-spatial and temporal scales, and social contexts.

While the ecological realm can be anthropogenic in a material sense, it is 'value-free' in the sense that it describes the biophysical flows of energy, matter and information. These flows can change across spatial and temporal scales and relate to the intensity and frequency of underlying ecosystem functions. It is, however dependent on human-value attribution that identifies those ecosystem attributes and functions which either enhance or reduce human well-being in given spatial and temporal contexts (Chan et al., 2012; Cumming et al., 2014; Reyers et al., 2013; Saunders and Luck, 2016). In our case, this human-value attribution can be understood as a measure of importance given to, or interest in, a particular phenomenon, be it ecological, economic or social.

2.4.3. The social realm

The social realm (Figure 2.1-B) relates to elements of human values, preferences, and principles (Chan et al., 2012), as well as human and institutional perception and behaviour produced in the complex, context-dependent dimensions of economics, politics, and culture. The latter define a demand for ecosystem services and an exposure to disservices, and the desirable or undesirable appropriation of the benefits or nuisances from ecosystems

(Spangenberg et al., 2014b). For example, landscape features can be perceived as cultural services (as defined in CICES; Haines-Young and Potschin, 2013) by some people - i.e. "*the landscape is beautiful*" - and as a cultural and aesthetic disservices (as proposed in Table 2.1) by other people - i.e. "*the landscape is ugly*". Such valuation depends on, among other things, the cognitive structure that people form through their experiences, individual cultures, preferences, principles, virtues, and norms, as well as the circumstances of the social, political and economic environment people are in (Brown and Westaway, 2011; Shackleton et al., 2016). Also, temporal advances in scientific, cultural, or generational knowledge might affect individual views of benefits or nuisances derived from the same ecosystem functions or attributes. For instance, environmental education can change the perception of particular species: bat species that were previously negatively perceived due to folklore (e.g. related to Dracula and feeding on human blood) become welcome after demystifying their negative impact and explaining their function in the ecosystem (e.g. pollinating tree fruits or predating mosquitoes; Kingston, 2015).

This operationalisation of the human valuation of ecosystem outcomes can be achieved through different approaches, ranging from monetary calculations of benefits or damages (TEEB, 2013), to the assessment of the willingness to do various things for a certain desire (Whelan et al., 2015), or the estimation of human happiness and satisfaction indices (Smith et al., 2013).

2.4.4. The social-ecological interface

The integration of ecological and social realms at the social-ecological interface (Figure 2.1-C) allows at distinguishing ecosystem attributes and functions without having a-priori values, thereby opening the concept to a broader assessment of benefits and nuisances.

We assume that ecosystem services and disservices are not entirely antagonistic, yet their beneficial or detrimental effects can be opposite to each other, i.e. benefits can express services or reduced disservices, and nuisances can express disservices or reduced services. For instance, human health can either benefit from several ecosystem services, such as food, pharmaceutics, and genetic materials; or from the mitigation of health disservices, namely the decrease of plant species with allergenic potential (reduced ecosystem disservices). Contrastingly, human health can be impacted by the decrease in quality and quantity of such ecosystem services (reduced services), as well as by the action of health disservices, namely food poisoning, ecosystem contamination, or diseases. Also, human safety and security can be positively influenced by the capacity of ecosystem services to regulate and mitigate events such as floods and by the decrease in the occurrence of certain security and safety disservices, such as the removal of trees prone to falling. The opposite is also possible: human safety and security can be minimised either by the loss of regulation and maintenance services, or by the enhancement of security and safety disservices (Table 2.1).

We emphasise that a clear separation of the social realm from the ecological realm is difficult since humans are part of both realms, with actions influencing the social-ecological interface. Social-ecological management, particularly feedbacks between ecological shifts and societal responses to social-ecological changes, may trigger, enhance or reduce either services or disservices. For instance, placing societal assets (such as houses) or ecological features (such as alien species) in systems prone to disturbance such as floodplains affected by floods or storms (or in fire-prone vegetation) may enhance potential nuisances; and the promotion of monocultures or the suppression of natural processes may trigger disservices and reduce services. Also, and especially under global change, services and disservices depend on the human capacity to adapt to/learn from ecosystem changes, i.e. the social-ecological memory from Nykvist and von Heland (2014). This can, for instance, favour the provision of certain ecosystem services, e.g. since people learned how to cope with changes; or the disappearance of services and emergence of ecosystem disservices, e.g. since people did not adapt to the novel ecosystem (see section 2.4.4).

Table 2.2. Examples of how different human well-being dimensions (based on the categories from Smith et al. (2013)) are affected by benefits, nuisances from reduced services (from ecosystem services, based on CICES), ecosystem disservices (based on Table 2.1), and social-ecological management.

Benefits to human well-being	**Nuisances for human well-being**		**Social-ecological management**
Ecosystem services	**Reduced ecosystem services**	**Ecosystem disservices**	
Health			
Provisioning services - quality of food and water, provision of pharmaceutics, genetic materials; *Regulating and maintenance services* - quality of water and food, regulation of climate, air quality, floods; control of pests and diseases.	Reduced mediation of waste, toxics and other nuisances (e.g. biochemical remediation by algae); Inadequate maintenance of physical, chemical, biological conditions (incl. atmospheric composition and climate regulation, and chemical condition of waters).	*Health disservices* - directly affecting human health, from pollution, poisoning and hygiene, to contamination, diseases and their spread (vectors such as mosquitos), and genetic resistance to pharmaceutics.	Mechanisms and organisations that regulate individual and societal health; Adequate resource management and technology that promote better conditions and quality of ecosystems.
Social cohesion			
Cultural services - physical and intellectual interactions with ecosystems, promoting a sense of place and shared experiences between communities and generations.	Disrupted mediation of mass, water, and gaseous flows, incl. natural events that could not be mediated by biota and provoke material damages (namely storms and floods) in communication infrastructures that bring people together.	*Material disservices* - biota damaging communication networks (namely rivers or roads) and lead to economic inequality (economic damages to poorer communities); *Leisure and recreation disservices* - unwanted ecosystem attributes that promote the lack of people connectedness.	Social norms that drive perception and promote social cohesion and equitability; Mechanisms that regulate the access to and enjoyment of ecosystems, creating opportunities for social interactions.
Education			
Cultural services - that allow intellectual development, cultural diversity, knowledge	Disruption of maintenance of physical, chemical, biological conditions, namely	*Cultural and aesthetic disservices* - repulsive feelings against species or	Education, finance and communication services;

systems, educational values and cognitive richness through nature.	unsuitable development of habitat nursery and animal reproduction.	ecosystem components and diminish the desire to learn with nature.	Creation of opportunities to access natural areas for teaching, researching, and transferring knowledge into arts and culture.
Safety and security			
Provisioning services - food and water security, energetic standards for safety; *Regulation and maintenance services* - integrity and quality of ecosystems, and the mitigation of unwanted ('natural') phenomena.	Disrupted mediation of flows (e.g. flood and storm protection, or mass stabilisation and control of erosion rates).	*Security and safety disservices* - related to people's perception towards fear; physical exposure to ecosystems that threaten human safety or facilities or enhance costs associated to natural damages; *Material disservices* - degradation of infrastructures that leads to economic inequality (economic damages to poorer communities).	Social mechanisms that provide protection, financial and social assistance in the case of damage; Integrated resource management and technology that regulate damages derived from wildlife or ecosystem processes.
Living standards			
Provisioning, regulating and maintenance, and cultural services - that provide a real or perceived increase in quality for daily living, including food, ornamental, economic outcomes, and ecosystem well-functioning.	Disrupted mediation of mass, water, and gaseous flows, including natural events that could not be mediated by biota and provoke material damages (namely storms and floods) in communication infrastructures.	*All types of disservices* - considered intrusive in daily life, from health disservices, to cultural and aesthetic disservices conflicting with people's beliefs, or material disservices promoting social costs.	Social mechanisms that promote wealth equality, improve living conditions, and that allow the maintenance and creation of green space, e.g. cost regulation for human health and infrastructures.
Leisure time			
Regulating and maintenance services - pleasant environment increasing the willingness for people to enjoy nature;	Inadequate maintenance of physical, chemical, biological conditions (e.g. lack or disruption of pest and disease control);	*Leisure and recreation disservices* - disrupted physical participation of people with nature, including the	Incentives for social interactions, networking and cohesion in nature (e.g. publicity);

Cultural services - opportunities for physical outdoor activities.	Reduced mediation of waste, toxics and other nuisances (e.g. bio-chemical remediation by algae).	opportunities for recreation and relaxation.	Social opportunities to access and interact with nature, incl. activities that enhance the perceived quality of recreational and aesthetic areas (e.g. footpaths, gardens).
Spiritual and cultural fulfilment			
Cultural services - physical, intellectual and spiritual interactions with nature, including aesthetic values, inspiration and cognitive development, and spiritual enrichment.	Disruption of development of habitat nursery and animal reproduction.	*Cultural and aesthetic disservices* - negative perception of nature on people's fulfilment, including cultural traditions, anxiety situations, fears, as well as unpleasant and repulsive due to beliefs (or past experiences).	Social opportunities for regulating inequality and improving cultural, educational and spiritual inclusion in heritage; Social mechanisms that protect and promote sacred and cultural aspects of ecosystems.
Connection to nature			
Regulating and maintenance services - sense of fulfilment towards nature outputs; *Cultural services* - intellectual and physical interactions with nature.	Inadequate maintenance of physical, chemical, biological conditions (e.g. lack or disruption of pest and disease control); Reduced mediation of waste, toxics and other nuisances (e.g. biochemical remediation by algae).	*Cultural and aesthetic disservices* - responsible for negative perceptions about nature experiences; *Leisure and recreation disservices* - disrupting physical willingness to connect with nature.	Policies and land use planning; community and faith-based initiatives; Social regulation of the condition of ecosystems, promoting access to nature and biodiversity, recreation and aesthetics.

2.4.5. The management hierarchy for ecosystem services and disservices

We argue that the notion of ecosystem disservices should account for the role of management in the consideration of human values attributed to ecosystems. In the past, the precautionary approach has often guided ecosystem services management, i.e. the assumption that anthropogenic disturbances of (natural) ecosystems should be avoided or reverted if possible. Such a framework might be too narrow when considering ecosystem disservices and the possibility of other ecosystem nuisances.

We expand a precautionary approach by suggesting a management hierarchy to guide social and technological actions. This hierarchy has been reported as being efficient in terms of policy development and management implementation for achieving ecosystem health and ecological sustainability (McKenney and Kiesecker, 2010; Tallis et al., 2015). We suggest expanding this framework so that it: first identifies and evaluates potential ecosystem services and disservices, which are relevant for a given social-ecological system; then, it protects and maximises ecosystem services, avoids and minimises disservices (while restores services); and finally, compensates and adapts to ecosystem disservices (Tallis et al., 2015; Figure 2.2).

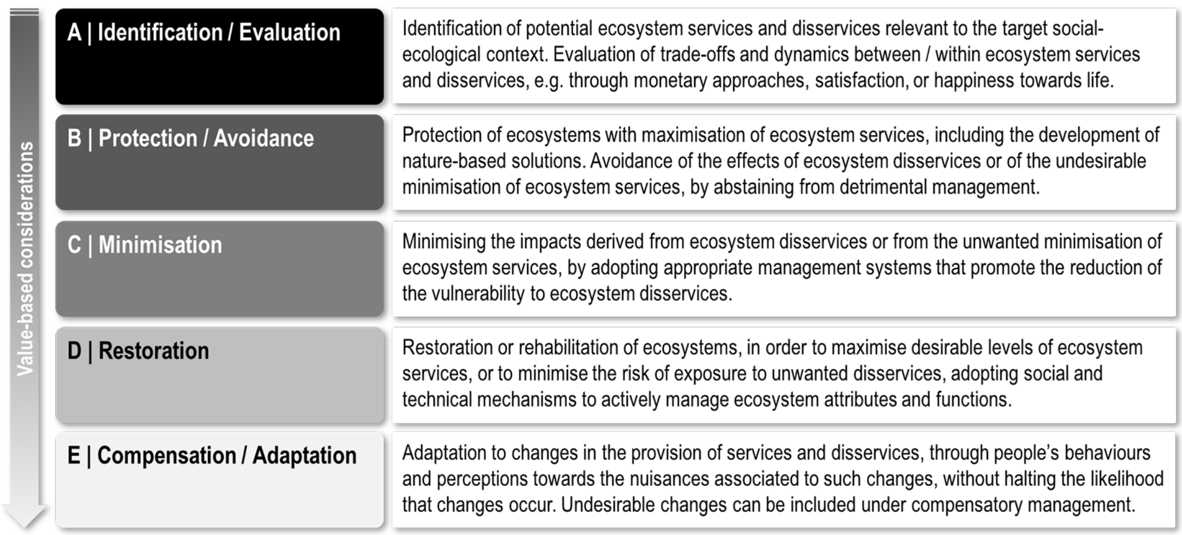

Figure 2.2. The management hierarchy proposed to identify strategic environmental management activities aimed at maximising ecosystem services and reducing ecosystem disservices.

Specifically, as a first step, the management hierarchy includes the identification and recognition of ecosystem outcomes, as well as the main trade-offs, synergies and dynamics between and within distinct ecosystem services and disservices. This evaluation may be conducted through several approaches, such as ecological and economic methods (TEEB, 2013), or by assessments of human satisfaction, preference, or happiness (Smith et al., 2013). As a second step, the hierarchy includes the protection of ecosystems or the maximisation of benefits from natural resources, including the development of nature-based solutions (Kabisch et al., 2016). Avoidance and minimisation strategies include actions for preventing and reducing impacts, whether they are derived from ecosystem disservices themselves or affect ecosystem services, e.g. by abstaining from detrimental management actions or pursuing technical solutions that allow for societal demands to be met while retaining ecosystem services supply. This can, for instance, be accomplished through technological development that minimises human dependency or overexploitation of services, i.e. the non-ecosystem services proposed by Cumming et al. (2014). Offsets of human management can be included under the restoration and compensatory mitigation of nuisances (Tallis et al., 2015), e.g. by adopting social and technological mechanisms to actively restore ecosystems in order to maximise desirable levels of ecosystem services' provision, or to minimise the risk of exposure to a specific ecosystem disservice (Biggs et al., 2012; Cumming et al., 2014; Reyers et al., 2013; Sagie et al., 2013). Adaptation considers strategies to cope with changes in the provision of ecosystem services (Biggs et al., 2012) and disservices, through actions that reduce these impacts without changing the likelihood that they will occur in the ecological realm, e.g. changes in people's behaviours and perceptions towards the nuisances associated to ecosystem disservices in the social realm.

Considering humans as simultaneous occupants of the social and the ecological realm means that context-dependent actions for the implementation of this hierarchy in our framework can be accomplished both through interventions that target the social realm (e.g. public awareness, governance dialogue, and the creation of social norms, mechanisms and opportunities) and the ecological realm (e.g. remediation of impacted areas by means of appropriate technology), depending on the multi-scale, -temporal, and -actor context (Tallis et al., 2015). For instance, while we can anticipate (identify) the possible occurrence of natural disasters, most often they can hardly be avoided. Yet, the protection and restoration of regulating services and the minimisation and adaptation to the nuisances from natural disasters can be accomplished though appropriate risk management, e.g. reduction of vulnerabilities and enhancement of resilience at specific social, political, and economic dimensions (Biggs et al., 2012). Table 2.2 exemplifies social-ecological management actions that can influence the amount of benefits and nuisances from ecosystem services and disservices to human well-being. These social-ecological actions are nevertheless dependent on value-based considerations: negotiations, discussions, debates, politics, and other values (and interests) involved in human choice towards evaluating and deciding which management activities are to be undertaken, e.g. to make priorities, or deal with conflicting views (Brown and Westaway, 2011; de Wit et al., 2001; McKenney and Kiesecker, 2010; Woodford et al., 2016).

2.5. THE ECOSYSTEM SERVICES AND DISSERVICES' TYPOLOGY AND FRAMEWORK ILLUSTRATED WITH PLANT INVASIONS

2.5.1. Plant invasions from an ecosystem disservices' perspective

Biological invasions are an interesting test case for our ecosystem disservices' typology and framework as negative outcomes on ecosystems and the need to manage ecosystems to minimise and adapt to such outcomes have long been discussed in the literature on the management of biological invasions (e.g. Brundu and Richardson, 2016; de Wit et al., 2001; Dickie et al., 2014; Funk et al., 2013). Here, we acknowledge alien plant species as those species that were introduced, accidentally or intentionally, by humans to new geographic areas. They may become invasive, i.e. spread from sites of introduction, and some may become abundant and cause diverse impacts on the environment or society (Richardson et al., 2011). Many invasive plant species have major impacts on biodiversity and ecosystem functioning (Eviner et al., 2012; Fenesi et al., 2015; Simberloff et al., 2013).

Based on our proposed disservices typology, we can identify different ecosystem disservices resulting from plant invasions: as providers of *health disservices* (e.g. through allergenic pollen transmission, Pyšek and Richardson, 2010; Schindler et al., 2015), *security and safety disservices* (e.g. creating fire occurrence in non-fire prone areas, Carruthers et al., 2011; Gaertner et al., 2014; Kull et al., 2011), *cultural and aesthetic disservices* (e.g. by forming monocultures perceived as unpleasant; Kueffer and Kull, 2017), or *leisure and recreation disservices* (by spoiling rivers for water activities in the wild; Figure 2.3a and b).

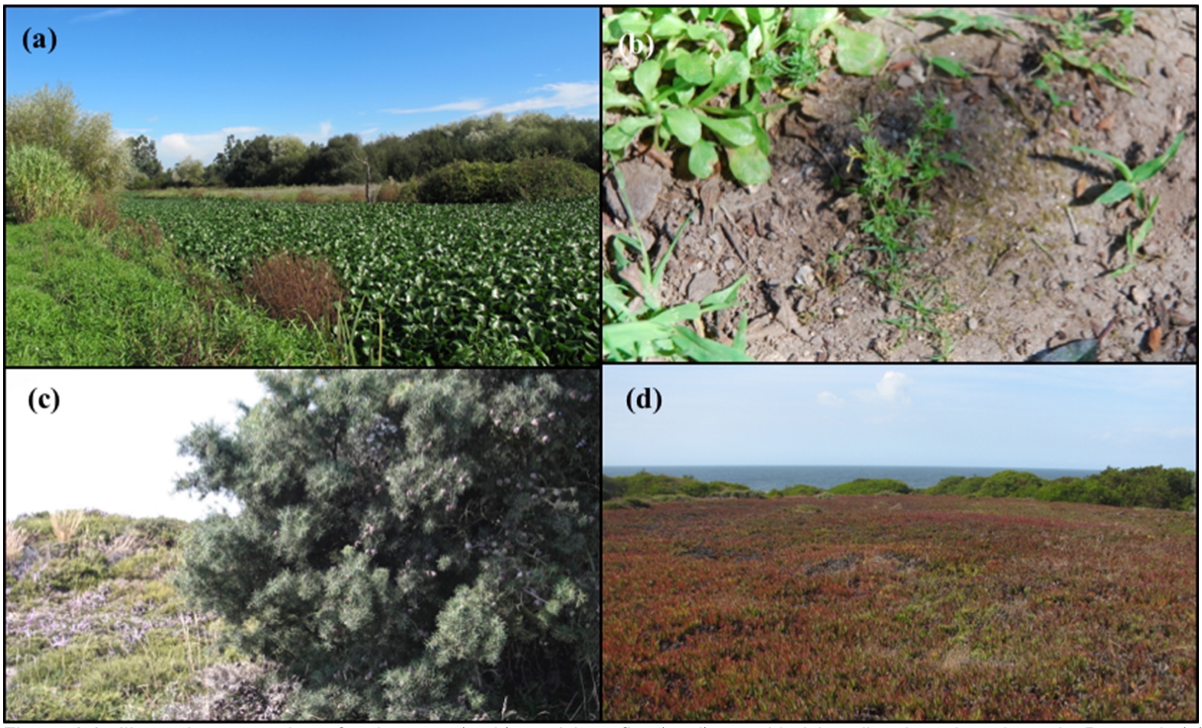

Figure 2.3. Illustrative examples of nuisances (a, b), and benefits (c, d), provided by alien invasive plants in Portugal: (a) *Eichhornia crassipes*, native to South America, which invades rivers, causing blockage and disrupting water transport and sports; (b) *Soliva sessilis*, native to South America, which invades lawns, causing physical injury to people and reduces recreation opportunities; (c) *Hakea sericea*, native to Australia, and introduced for afforestation goals and to be used by locals as fences; and (d) *Carpobrotus edulis*, native to South Africa, and introduced for ornamental uses and dune fixation in coastal habitats.

2.5.2. Plant invasions at the social-ecological interface

Although plant invasions can act as disservice providers (Shackleton et al., 2016), they can also provide important services (Figure 2.3c and d). Many alien plant species have been intentionally introduced to new areas to provide ecosystem services to individuals or groups of people, such as provisioning or aesthetics (Carruthers et al., 2011; Dickie et al., 2014; Kull et al., 2011), or to minimise the effects of a given disservices, such as pests. Many plant invaders are key resources in social-ecological systems around the world, especially in poor communities (Dickie et al., 2014; Kull et al., 2011), or in production systems (Koskela et al., 2014; van Wilgen and Richardson, 2014).

At the ecological realm, plant invaders can impact on several attributes and functions (see Table S2.2) that sustain the provision of wood and food, while at the same time changing the functioning or quality of other ecosystem functions and attributes, e.g. by reducing water quality and amount, or disrupting coastal sediment movement (Dickie et al., 2014; Gaertner et al., 2014; Koskela et al., 2014; van Wilgen and Richardson, 2014). The simple establishment of an invasive plant can contribute to both carbon sequestration and food provision in some spatial and temporal contexts (Dickie et al., 2014) or increase fire load and promote competition with native species in other contexts (de Wit et al., 2001; Fenesi et al., 2015). Also, changes in the biomass of invasive plants can contribute to ornamental enjoyment and spiritual perception (Carruthers et al., 2011; Shackleton et al., 2007) or result in the loss of a perceived wilderness character of landscapes and conservation areas (de Wit et al., 2001; Shackleton et al., 2007; see Table S2.2 for further examples).

The set of changes provoked by plant invasions are space and time dependent (Eviner et al., 2012): they relate with the stage of the invasion process (Simberloff et al., 2013). There is thus a complex interaction between the type and magnitude of invasion impacts depending on, among other things, the characteristics of the species, their invasive potential, the extent and time of invasion, and features of the invaded environment (Gaertner et al., 2014; Kueffer et al., 2013; Pyšek and Richardson, 2010; van Wilgen and Richardson, 2014). Ornamental plants are examples that provide benefits in private gardens but may cause nuisances once widespread in the wild (Dehnen-Schmutz et al., 2007).

The actual benefits (ecosystem services) or nuisances (ecosystem disservices and reduced services), emerging from the changes triggered by plant invasions in ecosystems, can only be recognised in the social realm. Specifically, the variety of values, socio-political conditions, perceptions, attitudes, knowledge and ideas attributed to plant invaders by humans (Kueffer and Kull, 2017) define the level and direction of impacts from these species. One of the most emblematic examples can be recognised in the genus *Acacia*. In Madagascar, *A. dealbata* can be exploited as a source of fuel wood and charcoal, contributing substantially to living standards and social cohesion, especially in poorer villages. In contrast, in Portugal, where it was introduced for afforestation and soil erosion prevention (Kull et al., 2011), *A. dealbata* is a widespread invader which decreases biodiversity and maximises fire hazard (Gaertner et al., 2014; Le Maitre et al., 2011). Another example is the mesquite (*Prosopis* spp.). The introduction of mesquite has been considered either as beneficial, due to the provision of food in Peru or wood in Kenya, or as source of security and safety disservices, leisure and recreation disservices and production disservices, such as physical injuries to humans in South Africa (Shackleton et al., 2015). Table 2.3 gives examples of benefits and nuisances promoted by plant invasions to human well-being.

Table 2.3. Examples of benefits, from ecosystem services (based on CICES), and nuisances from ecosystem disservices (based on Table 2.1) and reduced ecosystem services, promoted by plant invasions on human well-being dimensions (categories based on Smith et al. (2013)).

Benefits from plant invasions to human well-being	**Nuisances from plant invasions for human well-being**	
Ecosystem services	**Reduced ecosystem services**	**Ecosystem disservices**
Health		
Provisioning services: Blood sugar medicine from *Prosopis* species in South Africa (Shackleton et al., 2014); Several medicinal and curative products derived from *Eichhornia crassipes* in Bangladesh (Rana and Akhter, 2010); Styptics or astringents extracted from *Acacia mearnsii* (Kull et al., 2011; de Wit et al., 2001); Other products from *Acacia*, *Cinnamomum* and *Spathodea* species across the globe (Dickie et al., 2014).	Reduction of the provision of medicinal products through the elimination of other medicinal plants, by Australian *Acacia* species in South Africa, Portugal and Chile (Le Maitre et al., 2011).	*Health disservices*: Physical injury due to *Opuntia* thorns in South Africa (Shackleton et al., 2007); Myocardia or gastroenteritis associated to the consumption of flowers and seeds of *Ailanthus altissima* and *Robinia pseudoacacia*, and cardiac problems and poisoning from *Echium plantagineum* and *Rhododendron ponticum* (Pyšek and Richardson, 2010); Pollen allergy and (or) dermatitis caused by *A. altissima*, *Acacia dealbata*, *Ambrosia artemisiifolia*, *Cortaderia selloana*, *Heracleum mantegazzianum* and *Schinus terebinthifolius* (Pyšek and Richardson, 2010); Transmission of human parasites through invasive plants (Schindler et al., 2015).
Social cohesion		
Provisioning services: Exchange of *Opuntia ficus-indica* fruits, supporting community relationships and nurturing reciprocity in South Africa (Shackleton et al., 2007). *Cultural services:*	Removal of *Acacia* species in South Africa leads to social conflicts, decreasing social cohesion (Dickie et al., 2014).	*Cultural and aesthetic disservices:* Conflicts of interest between people (e.g. xenophobia, and conservationists versus land managers) due to *Acacia* and *Jacaranda* species in South Africa (Dickie et al., 2014; van Wilgen and Richardson, 2014); Conflicts over limited natural resources between communities in Ethiopia and Kenya due to *Prosopis* species (Shackleton et al., 2014);

Social equity associated to people's accessibility to *Acacia* species in South Africa, providing a sense of national symbolism (Carruthers et al., 2011); Poverty alleviation through employment, training and collaboration on managing *Acacia* species (Kullet al., 2011; McConnachie et al., 2013; Mugido et al., 2014; Shackleton et al., 2007).		Discrimination of people due to compromised selection of those benefiting from funding targeting invasive species management in South Africa (McConnachie et al., 2013). *Material disservices:* Blocked accessibility between humans within nature reserves in South Africa caused by *Opuntia ficus-indica* (Shackleton et al., 2007); Disruption of 'healthy country' including important cultural sites in Aboriginal Australia (Bach and Larson, 2017).
Education		
Cultural services: Opportunities for environmental education and training focused on management of *Acacia* (Carruthers et al., 2011; Mugido et al., 2014) and *Opuntia* species (Shackleton et al., 2007).	Disruption of personal identity due to misleading national symbolism of Australian and African *Acacia* species (Carruthers et al., 2011).	-
Safety and security		
Provisioning services: Financial security through cash income from *Opuntia* species in South Africa (Shackleton et al., 2007); Species as fence poles, namely *Acacia* and *Pinus* (Dickie et al., 2014). *Regulation and maintenance services:* Soil conservation, stabilisation and fertility, land reclamation, windbreaks against sandstorm, watershed protection, dune stabilisation, roads protection, from *Acacia* and *Pinus* species in Kenya, Madagascar, South Africa, New Zealand (Carruthers et al., 2011; de Wit et al., 2001; Kull et al., 2011; Rana and Akhter,	Compromised food security due to impacts on livestock health by Prosopis species in Kenya (Shackleton et al., 2014); Changes in fire and flood regimes promoted by *Acacia* species (Gaertner et al., 2014; Le Maitre et al., 2011; van Wilgen and Richardson, 2014); Modification of soil quality and promotion of soil erosion (de Wit et al., 2001; Funk et al., 2013; Shackleton et al., 2014).	*Security and safety disservices:* Harbouring of criminals in dense vegetation of *Acacia mearnsii* in South Africa (Shackleton et al., 2014); Species that promote fire hazard in non-fire prone areas (Carruthers et al., 2011; Gaertner et al., 2014; Kull et al., 2011; Richardson and van Wilgen, 2004).

2010; Richardson and van Wilgen, 2004; Shackleton et al., 2007, 2014).		
Living standards		
Provisioning services: Crops, fruits, honey, fuelwood, tannins, timber and pulp for paper, namely from *Acacia* spp., *Eriobotrya japonica*, *Ficus carica, Opuntia spp., Morus alba* and *Psidium guajava* (Carruthers et al., 2011; Dickie et al., 2014; Koskela et al., 2014; Kull et al., 2011; Le Maitre et al., 2011; Shackleton et al., 2007; van Wilgen and Richardson, 2014); Fodder for cattle from *Opuntia* species in South Africa (Shackleton et al., 2007). *Regulation and maintenance services:* Carbon sequestration and nitrogen fixation (de Wit et al., 2001; Dickie et al., 2014; Kull et al., 2011; Qiu, 2015; van Wilgen and Richardson, 2014), namely by *Acacia* species in Portugal (Vicente et al., 2013); Sand stabilisation or erosion control, especially in degraded areas by several tree species (van Wilgen and Richardson, 2014). *Cultural services:* *Acacia* species associated to heritage, religion, folklore, fairy tales, legends and associated rituals (Kull et al., 2011).	Depletion of water sources for both consumption and irrigation, promoted by *Acacia* and *Prosopis* species in South Africa, Portugal and Madagascar (Carruthers et al., 2011; Funk et al., 2013; Kull et al., 2011; Le Maitre et al., 2011; Shackleton et al., 2015; van Wilgen and Richardson, 2014; Vicente et al., 2013); Disruption of soil-nutrient cycling, carbon and nitrogen fixation (Gaertner et al., 2014; Qiu, 2015); Loss of land for cattle due to dense vegetation (Kull et al., 2011; Shackleton et al., 2015; van Wilgen and Richardson, 2014); Destruction of timber resources by competition with other tree species (Richardson and van Wilgen, 2004); Pest transmission to tree plantations, promoted by *Acacia dealbata* in Chile, *A. longifolia* in Portugal and *A. saligna* in South Africa (Koskela et al., 2014; Le Maitre et al., 2011);	*Health disservices:* Constipation caused by the ingestion of *Opuntia ficus-indica* fruits in South Africa (Shackleton et al., 2007). *Material disservices:* Blocked accessibility within lands due to *Opuntia ficus-indica* expansion (Shackleton et al., 2007). *Security and safety disservices:* Forests considered a security risk and used as latrine areas (Kull et al., 2011; Shackleton et al., 2007, 2014). *Cultural and aesthetic disservices*: '*Ugly landscapes*' dominated by *Acacia* species (Carruthers et al., 2011). *Leisure and recreation disservices:* Physical injury through contact with the plant spines from several invasive species (Pyšek and Richardson, 2010; Shackleton et al., 2007, 2014).

	Other disease transmission to livestock in Kenya (Shackleton et al., 2014).	
Leisure time		
Cultural services: Species introduced as shade trees providing opportunity for picnic grounds; e.g. pines in Cape Town (e.g. Pooley, 2014), *Eucalyptus* species in South Africa, *Pinus* species in New Zealand, and *Rhamnus* and *Salix* species in Australia (Dickie et al., 2014).	Degradation of recreational areas and loss of touristic experiences (de Wit et al., 2001; Le Maitre et al., 2011; van Wilgen and Richardson, 2014); Blockage of water bodies by *Acacia mearnsii* in South African rivers (Shackleton et al., 2014), and tracks due to impenetrable stands (Pyšek and Richardson, 2010).	*Leisure and recreation disservices:* Discomfort caused when barefooted people contact with the thorns of *Prosopis* species (Shackleton et al., 2007).
Spiritual and cultural fulfilment		
Cultural services: Encouraging native biodiversity conservation, due to the appearance of exotic unpleasant *Acacia* species (Carruthers et al., 2011); Spiritual and aesthetic values attributed to "plant of my ancestors", production of traditional wines and jams from *Opuntia* species in South Africa (Shackleton et al., 2007); Use of *Acacia mearnsii* for building traditional huts, sacred pool protection, firewood to support traditional ceremonies, rituals and celebrations in South Africa (Shackleton et al., 2007); Visual amenity, ornamental purposes and landscape re-green provided by invasive plants (Carruthers et al., 2011; Dickie et al., 2014; Koskela et al., 2014; Kull et al., 2011; Le Maitre et al., 2011; Shackleton et al., 2007);	Loss of sense of place and aesthetic values due to the presence of invasive species, such as *Acacia*, *Opuntia* and *Prosopis* species in South Africa and New Zealand (de Wit et al., 2001; Le Maitre et al., 2011; Shackleton et al., 2007); Threats to national pride by replacing native, emblematic species (Carruthers et al., 2011; van Wilgen and Richardson, 2014); Reduced cultural value of sacred pools due to the presence of *Acacia mearnsii* in South Africa (Shackleton et al., 2007).	*Cultural and aesthetic disservices:* Lack of beauty, art and fascination that humans experience in wild nature or historic landscapes related to the invasion by *Acacia* species (Carruthers et al., 2011).

Provision of a ‘sense of place’ in urban areas associated to *Jacaranda* species in South Africa and Pinus in New Zealand (Dickie et al., 2014).		
Connection to nature		
Regulation and maintenance services: Reduced harvesting pressure on native plants by the collection of *Acacia mearnsii* (Carruthers et al., 2011; Shackleton et al., 2007); Provision of food for native wildlife, protection from predators, increased species richness by invasive vegetation (Dickie et al., 2014; Koskela et al., 2014).	Genetic pollution, leading to the dilution and loss of unique diversity in the wild, mainly by tree invasions (Koskela et al., 2014; Le Maitre et al., 2011); Global erosion of biodiversity and habitats (Carruthers et al., 2011; Dickie et al., 2014; Kull et al., 2011; Le Maitre et al., 2011; Shackleton et al., 2016; van Wilgen and Richardson, 2014; Vicente et al., 2013).	*Cultural and aesthetic disservices:* Appearance of monospecific forests of *Acacia cyclops, A. longifolia,* and *A. saligna* in South Africa (Gaertner et al., 2014).

As for the social realm, the beneficial or detrimental effects that plant invaders have on well-being inevitably shift according to the temporal and geographical context, and overall institutional, political and technological context of the human society impacted by plant invasions. For instance, while the introduction of *Acacia mearnsii* could be considered to have benefitted the South African economy in the past (Kull et al., 2011; van Wilgen and Richardson, 2014), the use of more advanced technology in the production of chemical tannins in South Africa reduced the current demand for the species (Carruthers et al., 2011). Another example results from the attribution of financial incentives, such as carbon credits (especially under the Kyoto protocol), which justifies afforestation with alien conifers in New Zealand (Dickie et al., 2014). Also, the perception of plant invasions as providers of benefits or nuisances can shift depending of the state of knowledge: although people may enjoy the beautiful flowers of *Acacia dealbata* or *Carpobrotus edulis*, the public awareness of these plants as promoters of water depletion and soil erosion in their lands, may alter people's perception of this species from beneficial to problematic (Marchante et al., 2010).

2.5.3. Plant invasions and the management hierarchy

A particular challenge of managing invasions is that their effects, valuation and management options are tightly interlinked (Humair et al., 2014; Kueffer, 2013; Woodford et al., 2016). Caution is warranted as an invasive plant may provide benefits or nuisances in the social realm, without necessarily being considered an a-priori beneficial or detrimental asset in the ecological realm. For instance, people will value invasions differently depending on available management options and the capacity to use ecosystem services and mitigate disservices provided by them, at certain geographical and temporal contexts (Kueffer, 2013).

There are thus trade-offs and synergies between the beneficial and detrimental effects of invasions, and levels of acceptance of these differ between societal actors (Humair et al., 2014; Kueffer, 2013; Kueffer and Kull, 2017; Saunders and Luck, 2016; van Wilgen and Richardson, 2014). Consequently, managing ecosystem services and disservices resulting from invasions is often only possible at the social-ecological interface. At this interface, managing invasions (and further social-ecological challenges) could first rely on: (1) identifying specific situations - which ecosystem functions are being modified, at which level, and how irreversible these changes are; (2) considering ecosystem complexity - which potential ecological dynamics and feedbacks can be altered and in which direction of change; (3) realising opportunities in ecosystem service change - how to balance benefits and nuisances, considering the distinct measures of human valuation; and (4) accounting for multiple social-ecological dimensions - how can invasion outcomes be altered in time, at multiple spatial scales, and through management, learning and changing social perception.

The management hierarchy that we propose offers a general strategy for assessing ecosystem services and disservices in the light of invasions, in that it provides an objective foundation on which to justify decisions about maximising benefits and reducing nuisances for human well-being. These decisions should be considered based on human values and interests involved when deciding which management actions are to be implemented, e.g. through deliberation about conflicting views and priorities in invasion management (Bach and Larson, 2017; de Wit et al., 2001; Humair et al., 2014). Management actions need to be tailored for particular geographic locations and time periods because the balance of ecosystem services and disservices will be different for different geographic and social-ecological contexts (Kueffer and Hirsch Hadorn, 2008).

We highlight four sequential strategies to manage the invasion process of alien plants from an ecosystem services-disservices' perspective. The first strategy involves the identification and assessment of potential changes in the ecological realm, including trade-off analyses that address the balance of benefits and nuisances provided by invasions in specific social-ecological contexts. The second strategy involves mostly prevention and early-detection actions, through either the protection of pre-existing ecosystem functions, or the enhancement of ecosystem functions and attributes leading to benefits, and the avoidance of potential nuisances derived from alien/invasive plants (as ecosystem disservices). The third strategy focuses on the mitigation and rapid response to the minimisation of ecosystem services and maximisation of disservices. This can be exemplified by distinct actions focused on the treatment of the invader itself and its effects, e.g. through eradication, containment and habitat restoration and rehabilitation (Funk et al., 2013; Simberloff et al., 2013; van Wilgen and Richardson, 2014). The last strategy involves adaptation to the occurrence or expansion of invasive plants, either by recognising potential novel ecosystem services (benefits) or accepting transformations on ecosystem services (including reduced ecosystem services) and the emergence of disservices (nuisances). Examples from adaptation include the use of plant invaders for livelihoods, the harvesting of species for bioenergy goals and wood (Mugido et al., 2014), or their maintenance for carbon sequestration or landscape aesthetics (Dickie et al., 2014; Gaertner et al., 2016; Shackleton et al., 2007).

2.6. Conclusions

Ecosystem attributes and functions can contribute both positively and negatively to human well-being. We therefore clarify the role of ecosystem disservices in the context of ecosystem services; in particular, since the ecosystem service notion has become an additional argument for biodiversity conservation and environmental sustainability. We invite the adoption of a modified typology for integrating the terms of ecosystem services and disservices under a common framework that considers their relation to ecosystem functions, human wellbeing and feedbacks between human actions and ecosystem functioning at the social-ecological interface. We illustrate our suggestions with the case of plant invasions.

Our framework and application are underpinned by three important assumptions. The first is that the ensemble of attributes and functions in a given ecosystem (ecological realm) are intrinsically value-free. The benefits or nuisances derived from ecosystem services and disservices are, however, dependent on value attribution from individuals, groups of individuals and societies addressed by the social realm (Shapiro and Báldi, 2014). These are shaped by their specific economic, cultural, and political context. Second, the strong spatial, temporal and socio-economic context-dependency of ecosystem services and disservices may not allow for a universal typology and single delineation of services and disservices. Differences in perceptions by societal actors and human management may trigger, maximise or minimise the impacts from ecosystem services and disservices. In this sense, services and disservices are not necessarily antagonistic but complementary concepts, while their beneficial versus detrimental effects can be opposite to each other. Third, because of the influence of human actions, services-disservices are coupled concepts and should not be perceived as static entities in dynamic ecosystems. In this context, a management hierarchy may be useful for achieving the overarching goal of sustainability, accounting for social and technological mechanisms to prevent, reduce or restore desirable levels of ecosystem services, and to minimise the risk of or exposure to a specific ecosystem disservice. This human management perspective broadens the original focus of the ecosystem service-disservice notion. It elucidates the nature of beneficial flows from ecosystems to society and additionally accounts for the role of value attribution and ecosystem management in flows of ecosystem services and disservices.

We are concerned that misinterpretations of our framework may arise. It may be argued that the proposed framework opens the door for too much negotiation about conservation and environmental management goals and priorities by explicitly considering a symmetry between ecosystem services and disservices. Our attempt to develop a comprehensive framework that is applicable across a wide range of ecosystems and socio-ecological contexts might also be criticised as being too reductionistic. Instead we believe that our framework facilitates targeted discussions and deliberations about dynamics related to nature and humans and how to manage (social-)ecological systems, thereby providing the means to steer debates beyond simplistic good versus bad dichotomies that are currently part of many environmental management efforts (e.g. Gaertner et al., 2016; Woodford et al., 2016). In our view, it paves the way for improved ecosystem management that is tailored to particular social-ecological contexts. We do not suggest that less attention should be paid to the beneficial roles of ecosystems and biodiversity for human well-being. Rather, we hope that our approach widens the conceptual understanding of ecosystem functioning, thereby expanding the repertoire of actions to protect and sustainably manage ecosystems and the services they provide. Finding ways to accurately balance ecosystem services and disservices with feasible (e)valuations of costs and added values to humans is a major challenge. We call for more attention from scientists to broader social-ecological challenges. By advancing the thinking on ecosystem disservices and by acknowledging the pivotal role of humans in the ecosystem services arena we hope that academics and practitioners will explicitly adopt a more dynamic notion of service-disservice coupling in changing ecosystems to account for human action and management at the social-ecological interface.

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SUPPLEMENTARY MATERIAL I

Appendix A - Association among typologies of ecosystem services

Table S2.1. The typology of ecosystem services considered in the proposed framework, based on CICES (Haines-Young and Potschin, 2013) and its association with the MA classification (based on MAES, 2013; Harrison et al., 2014), with examples of ecosystem services (from MA, 2005; Haines-Young and Potschin, 2013).

CICES (division | group)	**MA**	**TEEB**	**Examples**
Provisioning	***Provisioning***	***Provisioning***	
Nutrition | Biomass	Food	Food	Cultivated crops, game, fisheries
Nutrition | Water	Fresh water	Water	Water collection or desalination for drinking
Materials | Biomass	Fibre Timber Genetic resources Biochemical resources Ornamental resources	Raw materials Genetic resources Medicinal resources Ornamental resources	Fibres, timber, pharmaceutics, genetic materials
Materials | Water	Fresh water	Water	Water for irrigation, or industrial use
Energy | Biomass	Ornamental resources	Ornamental resources	Wood fuel, biota for energy production
Regulation & maintenance	***Regulating*** ***Supporting***	***Regulating*** ***Habitat or supporting***	
Mediation of waste, toxics and other nuisances | Mediation by biota	Water purification and water treatment	Waste treatment (water purification)	Waste water cleaning, degrading oil spills by marine bacteria, (phyto)degradation
Mediation of waste, toxics and other nuisances | Mediation by ecosystems	Water purification and water treatment	Waste treatment (water purification)	Adsorption of heavy metals and organic compounds in ecosystems (both biotic and abiotic factors), green infrastructure to reduce noise and smells

Mediation of flows | Mass flows	Erosion regulation	Erosion prevention	Erosion flow protection
Mediation of flows | Liquid flows	Water regulation	Regulation of water flows Moderation of extreme events	Coastal flood prevention
Mediation of flows | Gaseous / air flows	Air quality regulation	Air quality regulation	Air ventilation from vegetation
Maintenance of physical, chemical, biological conditions | Lifecycle maintenance, habitat and gene pool protection	Pollination Primary production Nutrient cycling	Pollination Maintenance on lifecycle of migratory species (incl. nursery service) Maintenance of genetic diversity (gene pool protection)	Pollination, habitats for plant and animal nursery and reproduction
Maintenance of physical, chemical, biological conditions | Pest and disease control	Pest regulation Disease regulation	Biological control	Pest and disease control, including invasive species
Maintenance of physical, chemical, biological conditions | Soil formation and composition	Soil formation Primary production Nutrient cycling	Maintenance of soil fertility Maintenance on lifecycle of migratory species (incl. nursery service)	Maintenance of soil fertility, nutrient storage
Maintenance of physical, chemical, biological conditions | Water conditions	Primary production Nutrient cycling	Maintenance on lifecycle of migratory species (incl. nursery service)	Maintenance of chemical condition of fresh- and marine-water columns
Maintenance of physical, chemical, biological conditions | Atmospheric composition and climate regulation	Climate regulation	Climate regulation	Carbon sequestration, maintenance of climate and air quality, maintenance of atmospheric patterns
Cultural	***Cultural***	***Cultural***	
Physical and intellectual interactions with biota, ecosystems, and land-/seascapes | Physical and experiential interactions	Recreation and ecotourism	Recreation and ecotourism	Bird watching, snorkelling, diving, leisure hunting
Physical and intellectual interactions with biota, ecosystems, and land-/seascapes | Intellectual and representative interactions	Aesthetic values Cultural diversity Knowledge systems and educational values	Aesthetic information Inspiration for culture, art and design Information for cognitive development	Scientific, educational, heritage, cultural, entertainment, aesthetic subjects from nature

Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes | Spiritual and/or emblematic	Spiritual and religious values Cultural diversity	Spiritual experience Inspiration for culture, art and design	Emblematic and sacred plants and animal, holy places
Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes | Other cultural outputs	Knowledge systems and educational values	Information for cognitive development	Enjoyment from wild species, willingness to preserve nature

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Appendix B - Potential impacts caused by plant invasions in the ecological realm

Table S2.2. Examples of potential impacts caused by plant invasions in the ecological realm. Since the ecological realm is value-free we highlight main changes reported by invasive plant species on the distinct flows of the ecosystem (regulation, habitat, production and information), with potential benefits and nuisances that may be originated to well-being in the social realm. For simplicity, we followed the categories for ecosystem functions as proposed by de Groot et al. (2002) regulation, habitat, production and, information functions (see also van Oudenhoven et al., 2012).

Effects on ecosystem attributes and functions	Potential benefits	Potential nuisances
Related to the regulation capacity of ecosystems		
- Changes in net primary production (Dickie et al., 2014; Ehrenfeld, 2010; Pyšek and Richardson, 2010); - Changes in net N_2O and CH_4 fluxes (Qiu, 2015); - Biomass alteration (Le Maitre et al., 2011); - Changes in rainfall interception and transpiration (de Wit et al., 2001; Ehrenfeld, 2010; Levine et al., 2003); - Changes in fire load, soil water repellency and erosion (de Wit et al., 2001; Gaertner et al., 2014); - Changes in riverbank channelling (de Wit et al., 2001) - Changes in litter volume/decomposition rates (Ehrenfeld, 2010; Gaertner et al., 2014; Le Maitre et al., 2011); - Changes in nutrient pools (Ehrenfeld, 2010; Pyšek and Richardson, 2010).	- Increased carbon sequestration (Dickie et al., 2014); - Increased nitrogen fixation in nutrient-poor soils (Qiu, 2015).	- Channelling followed by slumping during floods (de Wit et al., 2001); - Increased soil erosion (de Wit et al., 2001); - Increased fire risk (de Wit et al., 2001); - Decreased native biodiversity (Gaertner et al., 2014; Downey and Richardson, 2016); - Promotion of pests and diseases (Le Maitre et al., 2011); - Diminishing water availability or distribution (Levine et al., 2003; Pyšek and Richardson, 2010).
Related to the habitat capacity of ecosystems		
- Increased biomass and vegetation height (Dickie et al., 2014); - Release of secondary compounds from roots (Gaertner et al., 2014); - Changes in soil nutrient levels (de Wit et al., 2001; Ehrenfeld, 2010);	- Habitat and food provision for wildlife, and protection from predators (Dickie et al., 2014); - Increased species richness (Gaertner et al., 2009).	- Competition with and destruction or damage of native plants (Gaertner et al., 2009, 2014; Downey and Richardson, 2016); - Suppressing of the germination of native seedlings (Gaertner et al., 2014);

- Competition for light and release of allelopathic compounds (Gaertner et al., 2014; Le Maitre et al., 2011; Pyšek and Richardson, 2010); - Changes in soil micro-environment (Gaertner et al., 2014); - Transfer of germplasm and hybridisation (Koskela et al., 2014; Pyšek and Richardson, 2010).		- Degradation of habitats, making them unsuitable for native biodiversity (de Wit et al., 2001; Ehrenfeld, 2010; Pyšek and Richardson, 2010); - Inhibition of soil biota functions and of mycorrhizal fungi (Gaertner et al., 2014); - Reduced species richness (Gaertner et al., 2009); - Genetic pollution and loss (Koskela et al., 2014; Le Maitre et al., 2011); - Increased greenhouse gas emissions (Qiu, 2015).
Related to the production capacity of ecosystems		
- Increased biomass and vegetation height (Dickie et al., 2014); - Increased reproductive performance (Schindler et al., 2015); - Changes in net primary production (Dickie et al., 2014; Ehrenfeld, 2010); - Production of nectar (Levine et al., 2003).	- Provision of fodder and shade for livestock (Dickie et al., 2014; Shackleton et al., 2007); - Increased pollination levels (Levine et al., 2003).	- Reduced grass cover for grazers (de Wit et al., 2001); - Decreased above- and belowground community production (Ehrenfeld, 2010); - Increased pollen production with health impacts (Schindler et al., 2015).
Related to the information flows of ecosystems		
- Changes in biomass, vegetation height and other morphological features (Carruthers et al., 2011; Dickie et al., 2014; Shackleton et al., 2007).	- Ornamental perception, enjoyment and spiritual identity (Carruthers et al., 2011; Shackleton et al., 2007).	- Reduced access for recreational activities (de Wit et al., 2001); - Loss of wilderness character of many rural landscapes and conservation areas (de Wit et al., 2001; Shackleton et al., 2007).

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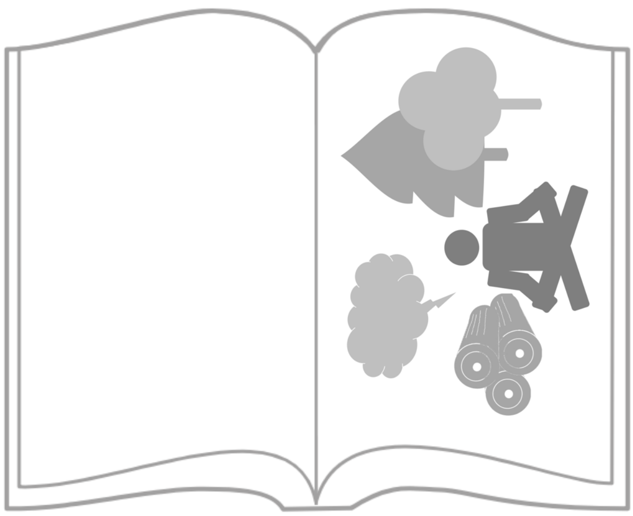
CHAPTER 3. REVIEW OF SOCIAL-ECOLOGICAL INTERDISCIPLINARITY IN INVASION SCIENCE



DISCLAIMER

This chapter is an original contribution of this thesis, published as a review paper in journal *Ambio* under the title "*The progress of interdisciplinarity in invasion science*". The paper had the contribution of the following authors:

Ana Sofia Vaz[a,b], Christoph Kueffer[c,d], Christian A. Kull[e,f], David M. Richardson[d], Stefan Schindler[g,h], A. Jesús Muñoz-Pajares[a], Joana R. Vicente[a,b], João Martins[a], Cang Hui[i,j], Ingolf Kühn[k,l,m], João P. Honrado[a,b]

[a]Research Network in Biodiversity and Evolutionary Biology, Research Center in Biodiversity and Genetic Resources (InBIO-CIBIO), Campus Agrário de Vairão, Rua Padre Armando Quintas, Vairão, Portugal;
[b]Faculty of Sciences, University of Porto (FCUP), Porto, Portugal;
[c]Institute of Integrative Biology, ETH Zurich, Universitätstrasse 16, Zurich, Switzerland;
[d]Centre for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, South Africa;
[e]Institute of Geography and Sustainability, University of Lausanne, Quartier UNIL-Mouline Bâtiment Géopolis, Lausanne, Switzerland;
[f]Centre for Geography and Environmental Science, Monash University, Clayton Vic 3800, Australia;
[g]Environment Agency Austria, Vienna, Austria;
[h]University of Vienna, Division of Conservation Biology, Vegetation and Landscape Ecology, Vienna, Austria;
[i]Centre for Invasion Biology, Department of Mathematical Sciences, Stellenbosch University, South Africa;
[j]Theoretical Ecology Group, African Institute for Mathematical Sciences, Cape Town, South Africa;
[k]Department of Community Ecology, UFZ - Helmholtz Centre for Environmental Research, Halle, Germany;
[l]Geobotany and Botanical Garden, Martin-Luther-University Halle-Wittenberg, Halle, Germany;
[m]German Centre for Integrative Biodiversity Research (iDiv) Halle-Jena-Leipzig, Leipzig, Germany.

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ABSTRACT

Interdisciplinarity is needed to gain knowledge of the ecology of invasive species and invaded ecosystems, and of the human dimensions of biological invasions. We combine a quantitative literature review with a qualitative historical narrative to document the progress of interdisciplinarity in invasion science since 1950. Our review shows that 92.4% of interdisciplinary publications (out of 9192) focus on ecological questions, 4.4% on social ones, and 3.2% on social-ecological ones. The emergence of invasion science out of ecology might explain why interdisciplinarity has remained mostly within the natural sciences. Nevertheless, invasion science is attracting social-ecological collaborations to understand ecological challenges, and to develop novel approaches to address new ideas, concepts, and invasion-related questions between scholars and stakeholders. We discuss ways to reframe invasion science as a field centred on interlinked social-ecological dynamics to bring science, governance and society together in a common effort to deal with invasions.

Keywords: Biological invasions; Interdisciplinarity; Non-native species; Scientometrics; Social-ecological research

3.1. INTRODUCTION

Humans influence processes that drive biological invasions by introducing species to new areas, facilitating their establishment and changing ecosystems in ways that enable the spread of these species (Kueffer and Hirsch Hadorn, 2008; Richardson et al., 2011; Kueffer, 2013; Hui and Richardson, 2017). Increasing globalisation has promoted the establishment and expansion of non-native species across the world (Hulme, 2009; Humair et al., 2015). Many introduced species are useful in new geographic areas, e.g. to provide resources or improve ecosystem services (Kull et al., 2011; Tassin and Kull, 2015; Vaz et al., 2017). However, a small proportion of non-native species becomes invasive (*sensu* Richardson et al., 2011), i.e. they spread, often becoming abundant, and in many cases have impacts on the environment and society. Some invasions contribute to major social-ecological changes - i.e. shifts in the state of ecosystems and coupled social systems - with positive or negative consequences for human values and welfare, such as those related to culture, health, and economy (Simberloff et al., 2013; Schindler et al., 2015; Kueffer and Kull, 2017; Vaz et al., 2017).

The social-ecological challenges arising from biological invasions have led to calls for insights from multiple disciplines (Kueffer and Hirsch Hadorn, 2008; Richardson, 2011a; Rotherham and Lambert, 2011; Kueffer, 2013; Matzek et al., 2013). Specifically, interdisciplinarity, at the interface of ecological and social sciences, is needed for understanding and managing invasions as an inherent social-ecological phenomenon (‘‘the human dimension’’, *sensu* McNeely, 2001). Such interdisciplinarity has been advocated to: (1) understand the multiple ecological and social drivers of invasions (Kueffer, 2013); (2) clarify social conflicts, interests, values, perceptions, and attitudes associated with non-native and invasive species (Larson, 2005; Estevez et al., 2014; Humair et al., 2014a; Kueffer and Kull, 2017); and (3) improve tools and strategies for management and policy (Kueffer and Hirsch Hadorn, 2008; Matzek et al., 2013; Head et al., 2015; Essl et al., 2017).

Invasion science, here understood as ‘‘*the study of the causes and consequences of the introduction of organisms to the areas outside their native ranges*’’ (Richardson and Ricciardi, 2011, p. 1461), combines interests from multiple disciplines to focus on e.g. species transportation, establishment and spread, biological interactions, and invasion costs and benefits to human systems (Richardson, 2011a; Essl et al., 2017). The pivotal role of (interdisciplinary) social-ecological approaches in invasion science has already been recognised (Kueffer and Hirsch Hadorn, 2008; Richardson, 2011a; Estevez et al., 2014; Head et al., 2015; Courchamp et al., 2017), specifically by economists, geographers, historians, philosophers, politicians, and sociologists (e.g. Larson, 2005; Carruthers et al., 2011; Hattingh, 2011; Kull et al., 2011; Rotherham and Lambert, 2011; Head and Atchison, 2015). Contributions from these scholars call for the elucidation of feedbacks between ecological and social drivers (Kueffer, 2013; Matzek et al., 2013), and the valuation of invasion effects which are co-produced by society, scientific facts, and cultural norms (McNeely, 2001; Hattingh, 2011; Kull et al., 2011; Estevez et al., 2014; Jeschke et al., 2014; Tassin and Kull, 2015; Essl et al., 2017; Kueffer and Kull, 2017). Other scholars have also focused on the role of societal beliefs, perceptions, memory, and cultural aspects related to non-native and invasive species that shape human attitudes, and therefore decisions relating to the management of these species (e.g. Carruthers et al., 2011; Estevez et al., 2014). Consequently, issues such as what constitutes a native or non-native species, whether a species is considered good or bad, and subsequent conservation and management positions (e.g. Rotherham and Lambert, 2011) are still debated amongst experts from different disciplinary backgrounds (Larson, 2007; Carruthers et al., 2011; Brunel et al., 2013; Humair et al., 2014a).

Given the growing appeal of interdisciplinarity, experts have called for a reframing of invasion science as a problem-oriented and multidisciplinary science, rather than a purely ecological science (Kueffer and Hirsch Hadorn, 2008; Kueffer, 2013; Estevez et al., 2014; Head et al., 2015; Essl et al., 2017). As in the case of other environmental challenges (Liu et al., 2007; Larson, 2011; Tengo et al., 2014; Rissman and Gillon, 2016; Bennett et al., 2017), a social-ecological lens can help to reframe invasion science (e.g. Larson, 2007; Kueffer, 2013; Matzek et al., 2013; Tassin and Kull, 2015) by better accounting for social-ecological feedbacks that mediate the dynamics and valuation of biological invasions (Kueffer, 2013; Kull et al., 2013; Head et al., 2015). A social-ecological perspective is particularly expected to improve the effectiveness of invasion science for management (e.g. McNeely, 2001; Kueffer and Hirsch Hadorn, 2008; Matzek et al., 2013; Tassin and Kull, 2015; Woodford et al., 2016; Hui and Richardson, 2017). Among other things, it is hoped that more robust social-ecological perspectives will help informing options for management at different stages of invasions (Kueffer and Hirsch Hadorn, 2008; N'Guyen et al., 2016; Essl et al., 2017). Human perception, culture, attitudes, ethics, actions, and adaptive learning-based approaches in invasion management can differ depending on the invasion stage, for example introduction versus spread phases (Rotherham and Lambert, 2011; Heger et al., 2013; Simberloff et al., 2013; Tassin and Kull, 2015; Chaffin et al., 2016).

Despite the recognition of a need for more cross-cutting collaborations, an overview of the extent of interdisciplinarity in invasion science is lacking. The first requirement to achieve such an overview is a thorough review of the state of interdisciplinarity in the field, based on published literature. Previous studies have reviewed the ecological literature (e.g. Davis et al., 2001; Davis, 2011), as well as the social and interdisciplinary literature in invasion science (McNeely, 2001; Kueffer and Hirsch Hadorn, 2008; Richardson, 2011a; Kueffer, 2013; Estevez et al., 2014; Kueffer and Kull, 2017). However, a broader quantitative assessment is still missing.

This paper examines the extent to which interdisciplinarity has featured in research addressing biological invasions over the last half-century. We begin with a quantitative analysis of interdisciplinary research in invasion literature, focussing on the integration of ecological and social sciences. Concurrently, we present a qualitative narrative of documented milestones of the progress of invasion science. We analyse the way in which social-ecological approaches in invasion science have been conceptualising: (1) how the causal influences between the social system and the invasion process (and vice versa) are described; (2) how impacts are characterised (anthropocentric versus ecocentric); and (3) whether research is focused on understanding causal relationships, valuation, or management support. We further investigate which stages of the invasion process and management strategies have been addressed from a social-ecological perspective. Finally, based on our quantitative review and temporal narrative of invasion science, we suggest avenues for fostering progress and adjusting the course of invasion research by reframing research questions through an explicit interdisciplinary and social-ecological approach.

3.2. A QUANTITATIVE REVIEW OF INTERDISCIPLINARY IN INVASION LITERATURE

3.2.1. Literature search

Following Richardson et al. (2011), we consider non-native species as those that were introduced (accidentally or intentionally) by humans to new geographic areas, and invasive species as non-native species that spread, sometimes becoming abundant and leading to major impacts on the environment or society. A literature search on non-native/invasive species was conducted using the ‘‘ISI Web of Science’’ core collection (ISI WOS; http:// webofknowledge.com/). Since we were interested in all relevant research related to biological invasions, we compiled a list of terms related to the main keyword ‘‘biological invasions’’ (Table

S3.1). The time span of our search was 1950-2014, corresponding to the period when the systematic study of invasive species began, after the publication of Elton's (1958) book (Richardson and Pysek, 2008; Hui and Richardson, 2017). Searches were conducted between February and September 2015. The records retrieved by the search (total number, n = 23640) were subjected to exclusion criteria to eliminate irrelevant information (e.g. topics such as invaders from outer space; see Table S3.2). These criteria were applied individually by checking the title and keywords of each record.

3.2.2. Records classification and analytical framework

A four-step analytical framework was applied to the final dataset (n = 23390), the aim being to classify the records according to their disciplinary and social-ecological scope (Figure 3.1).

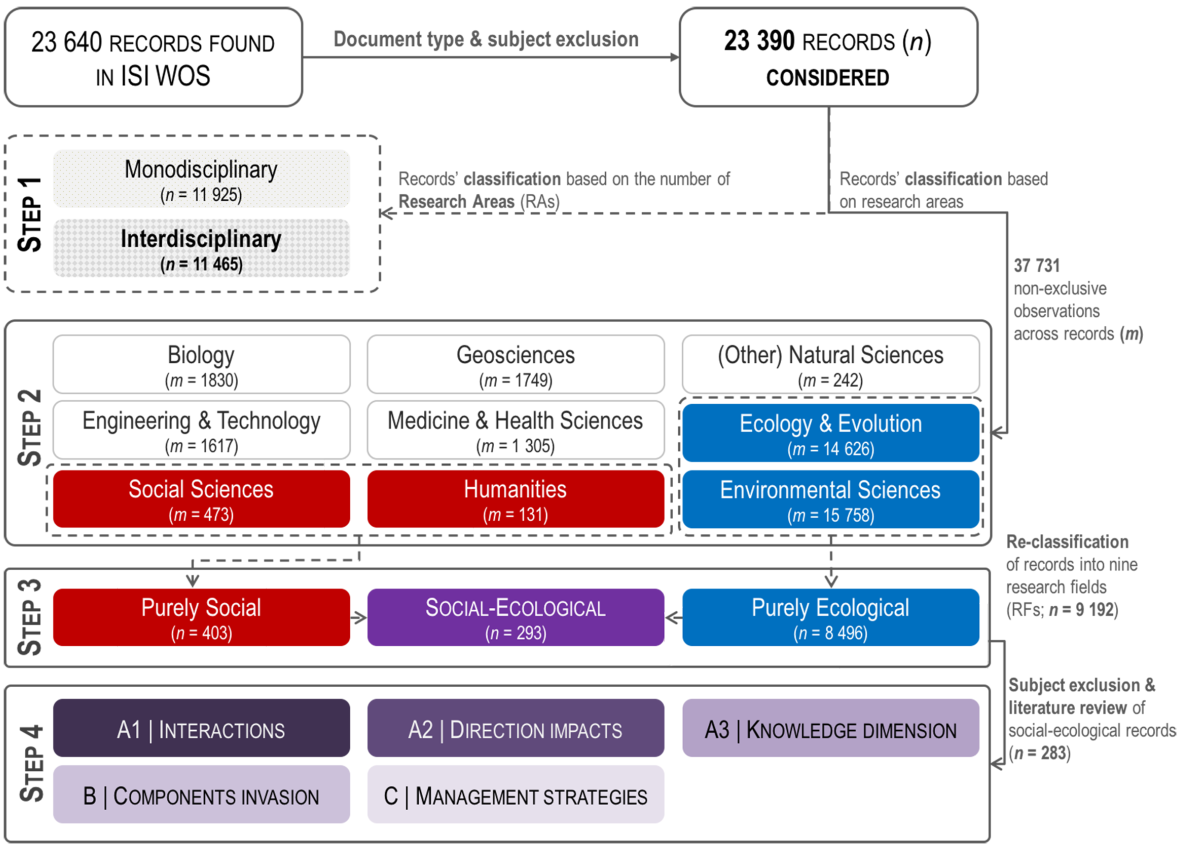

Figure 3.1. Analytical framework adopted to determine the incidence of interdisciplinary and social-ecological research in the literature of biological invasions. Search engine: ISI Web of Science (WOS), time span of the search: 1950-2014. The framework included four steps: in Step 1, we classified each of the 23390 records as either inter- or monodisciplinary based on the number of research areas (RAs) assigned to each record by WOS; in Step 2, we classified each RA and its respective records into one of nine broader research fields (RFs); in Step 3, we aggregated the categories determined in Step 2 and classified each record as either social, ecological, or social-ecological; in Step 4, we analysed all records that were classified as social-ecological in more detail considering several focal questions (see Table 3.1 for more information).

In the first step, we classified each record based on the number of different “Research Areas” (hereafter RAs) according to ISI WOS, as either interdisciplinary (attributed to at least two RAs) or monodisciplinary (Rafols et al., 2010; Stock and Burton, 2011). A total of 110 RAs was retrieved. The full list of RAs is shown in Table S3.3. The RAs considered here correspond to the scientific disciplines attributed to each individual record by ISI WOS. These categories are widely applied in scientometrics for the evaluation of interdisciplinarity research (Porter and Rafols, 2009; Rafols et al., 2010; Wagner et al., 2011). We are aware that pre-existing categorisations have limitations for measuring interdisciplinarity (e.g. due to a lack of consensus regarding the accuracy of the classification, or because one RA is nested within another RA). However, the system is well-established, improving our ability to compare classifications across large areas of science and with thousands of studies (Rafols et al., 2010). We are also aware that when disciplines join forces to solve a common problem, other terms are used (i.e. cross-, multi-, inter-, trans-, supra-disciplinarity) which also have slightly different meanings. Since it was beyond the scope of our study to explore differences among disciplinarity concepts, we adopted “interdisciplinarity” in the bibliometric portion of this study as a lowest common denominator umbrella term for designating a research publication that draws on, or involves from, more than one discipline (see e.g. Stock and Burton, 2011).

In the second step, we grouped RAs into nine broader research fields. The classification into research fields was conducted by our interdisciplinary team, supported by the description of RAs provided by ISI WOS and following the works of Leydesdorff and Rafols (2009), Porter and Rafols (2009), Rafols et al. (2010), and Wagner et al. (2011). We are confident that this classification represents the most intuitive combinations of RAs in the literature of biological invasions, while it facilitates the disclosure of the set of RAs retrieved by our search. (This is the reason why the fields of ecology, environment, biology, and (other) natural sciences were considered as separated research fields, whereas the broad research fields of social sciences and humanities were not subdivided; see Table S3.3 for details on research fields and categorisation).

In the third step, we grouped the RAs into two broad categories: ecological/environmental, and social/human. For the ecological/environmental category, we combined *Ecology & Evolution* with *Environmental Sciences*. For the social/human category, we combined *Social Sciences* and *Humanities*. The remaining scientific fields were not so considered, since our main focus was on ecological, social, or social-ecological records. We then classified each record as purely ecological (i.e. records that only comprise RAs categorised as ecological/environmental), purely social (i.e. records that only comprise RAs categorised as social/human), or social-ecological (i.e. records which comprise RAs from both ecological/environmental and social/human categories). In the final step, we analysed all records that were classified as social-ecological (n = 293 out of 23390). Each record was reviewed to confirm its social-ecological scope by screening the title, keywords, and abstract. After removing unsuitable articles, the full text of the final set of records (n = 283) was analysed to answer a set of focal questions related to our objectives (Table 3.1).

Table 3.1. Focal questions and categories considered in Step 4 of the framework, with a detailed description and references.

Codes	Description
STEP 4A. Social-ecological approaches	
A1. What is the main direction of influence between the social system (S) and the invasion (I) process? *(adapted from Binder et al., 2013)*	
Social→Invasion (S→I)	The social system drives the invasion process
Invasion→Social (I→S)	The invasion process influences the social system
Invasion↔Social (S↔I)	There is reciprocity between the two systems
A2. What is the main direction of impacts provoked by the invasion process? *(adapted from Binder et al., 2013)*	
Anthropocentric	The invasion process provokes impacts (partially/totally) on the social system
Ecocentric	The invasion process provokes impacts exclusively on the ecosystem
A3. Which knowledge dimension does the study produce? *(Kueffer and Hirsch Hadorn, 2008)*	
Systems knowledge ('causes')	Oriented towards analysing and improving the causal understanding of the invasion process
Target knowledge ('valuation')	Oriented towards clarifying conflicts of interests and values, including peoples' perceptions, valuations and conceptualisations
Transformation knowledge ('solutions')	Oriented towards improving or avoiding a particular situation related to the invasion process
STEP 4B. Invasion process	
B. Which stages of the invasion process are studied? *(adapted from Van Wilgen et al., 2014)*	
Introduction	Focuses on the pathways of species introduction from one geographical region to another
Establishment	Focuses on the determinants of success of species establishment
Expansion	Focuses on the patterns and mechanisms of species expansion
Dominance	Focuses on patterns and processes related to invaders that have become dominant in an invaded area, including impacts and management
Stage independent	Studies that do not specify the stage of the invasion process, mostly because they address the invasion process as a whole
STEP 4C. Management type	
C. Which strategies of invasion management are considered by the study? *(Van Wilgen et al., 2014)*	
Prevention	Focuses on preventing the introduction of new invasive species (including risk assessments of source areas, spread pathways, and species characteristics)
Monitoring	Focuses on mapping, assessing and monitoring the distribution and impacts of invasive species

Mitigation	Focuses on reducing the (likelihood) of impacts of an invasive species, including containment of further spread and eradication
Adaptation	Focuses on dealing with and tolerating impacts (including tolerating species or using them, e.g., for timber, medicinal or ornamental purposes)
No management	There is no focus on the management of invasive species
Unspecified	The study does not specify the management type

3.2.3. Interdisciplinarity analysis

The level of interdisciplinarity in our dataset was first illustrated through network plots (Butts et al., 2015), and then measured based on the declining rate of zeta diversity (Hui and McGeoch, 2014).

First, interdisciplinarity was visualised using network plots for each individual year (Rafols et al., 2010; Wagner et al., 2011). For each network produced, a given RA is represented by a node (or circle), and the relationship between a given combination of two RAs is represented by a connecting line. The thickness of the line in the network represents the number of records which are classified under both RAs (Rafols et al., 2010). Network plots were constructed using the network package (Butts et al., 2015) implemented in R (R Core Team, 2014).

Next, the level of interdisciplinarity was quantified using metrics of the declining rate of zeta diversity, which expresses the number of RAs shared by multiple records (Hui and McGeoch, 2014). Specifically, zeta diversity of order 1 depicts the average number of RAs per paper; zeta diversity of order 2 depicts the average number of RAs shared by two papers; zeta diversity of order *n* depicts the average number of RAs shared by *n* papers. Because RAs shared by *n* papers will also be shared by *n*-1 papers, zeta diversity declines monotonically with its order, either exponentially or following a power law depending on whether the RAs are randomly assigned to each paper or inherently different among papers. Since all our cases followed a power law zeta diversity decline, we chose to use the absolute exponent of the power law as a metric of interdisciplinarity, calculated based on non-linear regression for zeta diversity of order 1-5 for a focal year. A low absolute exponent represents a higher number of RAs shared by a large number of papers (and thus higher interdisciplinarity), and a lower number of RAs exclusive to selected papers especially those with fewer RAs. The rate of zeta diversity was calculated for the whole dataset (Step 1) and for each category of records classified as social, ecological, or social-ecological, at an annual pace (Step 3). In Step 3, due to the small number of records found before 1990, the rate of zeta diversity was computed by pooling the entire research during the period 1950-1990, and then at an annual pace until 2014. Computations were implemented in the package zetadiv (Latombe et al., 2015) available in R software (R Core Team, 2014). Results are presented as line or column plots.

3.3. An historical overview of invasion research

The history of invasion science has been discussed previously (Davis et al., 2001; Davis, 2006; Kueffer and Hirsch Hadorn, 2008; Richardson and Pysek, 2008; Chew and Hamilton, 2011; Hobbs and Richardson, 2011; Simberloff, 2011; Hui and Richardson, 2017). We provide quantitative data on this historical overview, suggesting that the invasion literature showed an exponential increase since the 1980s, with the steepest slope after 1990 (Figure 3.2a).

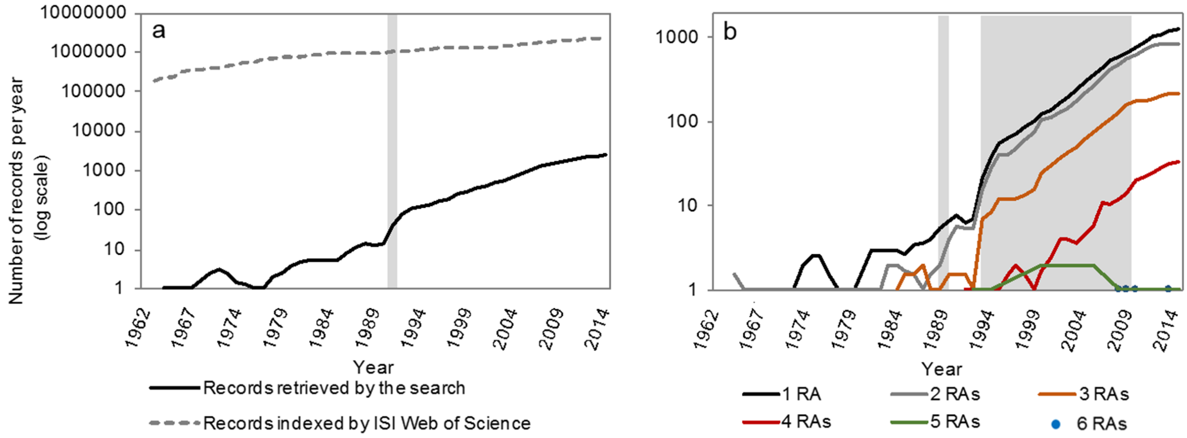

Figure 3.2. The number of records retrieved by the search on invasion literature in ISI Web of Science (WOS) from1950 to 2014 (smoothing curves showing averages for 3-year time periods), with the total number of records covered in WOS shown for comparison (a), and the number of different research areas (RAs) attributed to each individual record (b). Time periods discussed in detail along the text are highlighted with a light grey colour. Values in the y-axis are expressed in a logarithmic scale.

We recognise that there has been occasional interest in non-native species and their effects on ecosystems since at least the 1700s (e.g. Curtis, 1783; Watson, 1847). However, the book by Elton (1958) on *The Ecology of Invasions by Animals and Plants* is generally considered as the beginning of the systematic scientific study of biological invasions (Richardson and Pysek, 2008; Richardson, 2011b). Elton's book brought together subjects, including ecology, evolution, biogeography, biological conservation, and social sciences, thereby envisioning an interdisciplinary scope for invasion science (Richardson and Pysek, 2007). Despite this milestone, few publications on invasions appeared before the 1970s (see also Davis, 2006; Lockwood et al., 2007; Richardson and Pysek, 2008; Estevez et al., 2014; Hui and

Richardson, 2017). This apparent lack of interest contrasts with the considerable advances made in ecology in general during this period, including the development of ideas and work in community ecology that were inspired by Elton's book. An explanation for this time lag may be that invasions were not yet widely considered a major global threat and therefore did not receive much attention (Richardson and Pysek, 2007, 2008; Richardson, 2011b; Hui and Richardson, 2017). Conservation and environmental problems were increasingly recognised in the 1970s and 1980s as topics within ecology. Population biologists applied their new concepts to the spread of non-native diseases and pests in ''natural ecosystems'' (e.g. Krebs, 1972), and more generally in biodiversity conservation (Stork and Astrin, 2014). The field of restoration ecology (Zhang et al., 2010), and research relating to global environmental change (Li et al., 2011) has also grown rapidly since 1980.

The rapid, tenfold acceleration of publications on invasions in the 1990s (Figure 3.2a) can, however, not be explained by this general trend alone. Rather, this increase might reflect the growing interest of academics in biological invasions (e.g. the *Third International Conference on Mediterranean Ecosystems* in Stellenbosch, South Africa, in 1980; Richardson, 2011b) and the impact of a major international SCOPE research program on biological invasions in the late 1980s (Drake et al., 1989; see also Richardson and Pysek, 2008; Richardson, 2011b; Simberloff, 2011; Hui and Richardson, 2017). As biological invasions constituted a new topic for research assessment and publication (namely on islands; Vitousek, 1988; Lovei, 1997), rapid institutionalisation took place both in science (e.g. through the launching of specialised journals like *Diversity and Distributions* and *Biological Invasions*, in 1998 and 1999, respectively), policy (e.g. through legislation like the *Convention on Biological Diversity*, the Bern Convention, US executive orders, EU regulations), and in publicly funded programs (e.g. DAISIE, GISP; Davis, 2006; Kueffer and Hirsch Hadorn, 2008; Brunel et al., 2013; Hui and Richardson, 2017). Thus, feedbacks between funding of scientific consortia and scientific and public interest might have maintained the further growth of the field with increasing publication and citation rates. The increased prominence of issues relating to invasions also ensured the integration of invasion science in wider interdisciplinarity perspectives (e.g. through the *Millennium Ecosystem Assessment* and the *Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*; Matzek et al., 2013), including recognition of human-mediated introductions (Lovei, 1997). Additionally, invasions remained a topic of interest for both basic (Sax et al., 2007) and applied research, for instance in restoration ecology (Hobbs and Richardson, 2011).

The growth of invasion science can thus be seen as a paradigmatic case of the adoption of a new issue in environmental research. Within 60 years, an issue that was not widely recognised as such has become one of the most prominent topics in environmental research and conservation policies. Decisive moments that explain the trajectory include: (1) the novel and broad conceptualisation by Charles Elton; (2) growing scientific and societal appreciation of conservation issues starting in the 1980s; (3) targeted funding of large international research consortia, especially the SCOPE program in the 1980s, which led to a rapid growth and internationalisation of the issue; (4) positive feedback between growing scientific and political interests focusing on the negative values attributed to invasive species, leading to increasing institutional support; and (5) the solid grounding of the research in ecology that ensured an ongoing and growing interest of basic research in the field.

3.4. THE PROGRESS OF INTERDISCIPLINARITY IN INVASION RESEARCH

Despite the interdisciplinary scope of the field, and an oft-stated belief that interdisciplinarity is essential for addressing global, social-ecological challenges, especially in invasion science (Lockwood et al., 2007; Kueffer and Hirsch Hadorn, 2008; Kueffer, 2013; Estevez et al., 2014), our results suggest that the rise of interdisciplinarity only weakly followed the growth of the field (Figure 3.2b). In fact, our quantitative review shows that more than half of current invasion literature (ca. 51.0% out of 23390 publications) comprises monodisciplinary records. The remaining 49.0% of records that are classified as interdisciplinary include two (78.0%), three (19.0%), four (2.0%), or more (1.0%) RAs. Monodisciplinary records cover 60.0% of journals retrieved by our search (out of 1737 journals). Interdisciplinary records with two RAs cover 31.0% of journals; records with three or more RAs concern 8.0% and 1.0% of journals, respectively.

Our quantitative review shows that between 1970 and 1990s, interdisciplinary collaborations were largely confined to interactions between disciplines within the natural sciences, and more specifically within the fields of ecology and environmental sciences (Figure 3.3; see also Supplementary video and Figure S3.1). Such work typically focused on issues pertaining to forestry, agricultural pests, fish and game management, livestock diseases, and threats to wildlife (Davis, 2006; Lockwood et al., 2007). An example found in our literature search is the study by Mann (1979) which reviewed the deliberate introduction of non-native shellfish, mentioning the introduced species and the consequences of those introductions for mariculture.

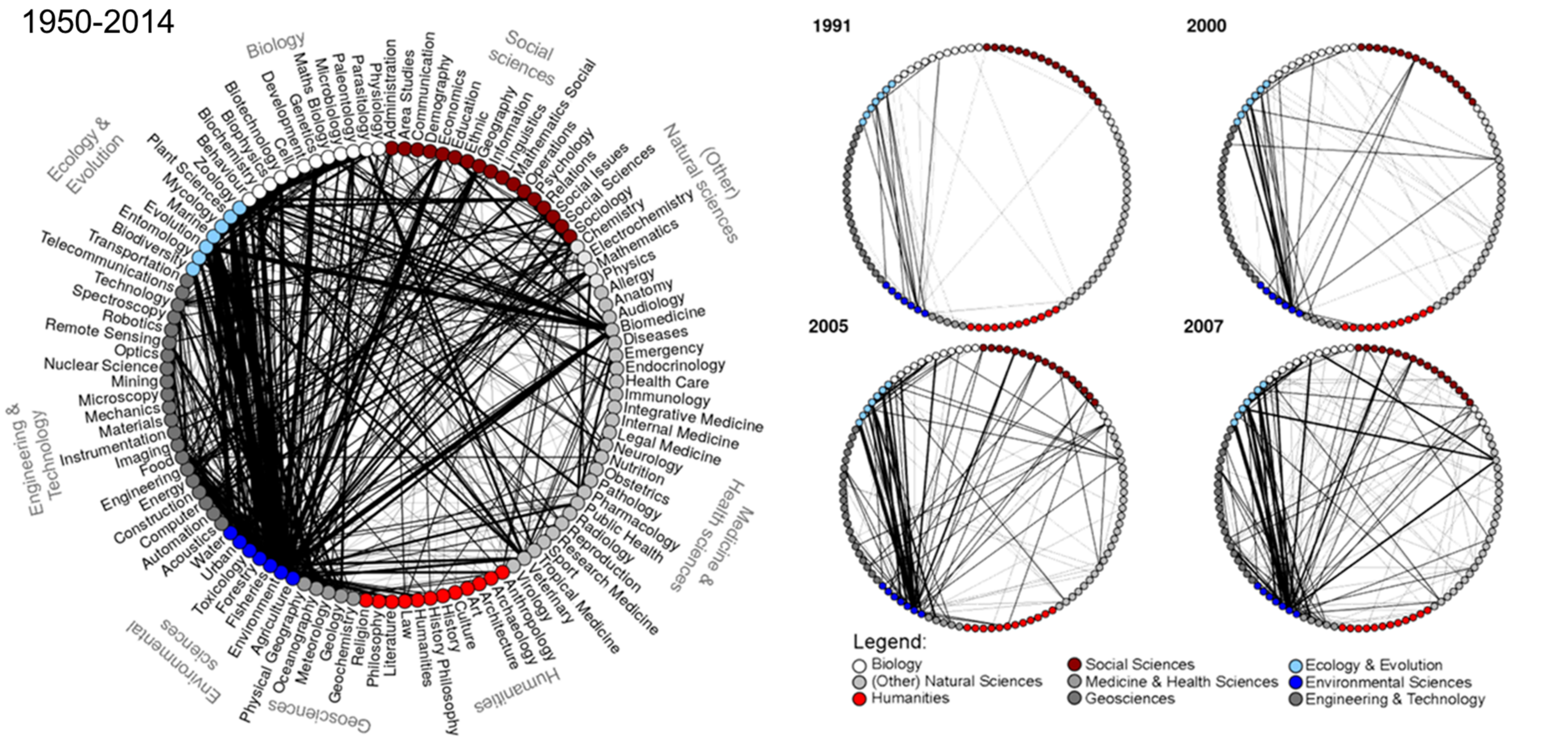

Figure 3.3. Network plots showing interdisciplinarity in invasion research for the period 1950-2014, and for the years 1991, 2000, 2005, and 2007, representative of the main transitions between the 1990s and 2000s (i.e. an increase in complexity of the combination of research areas, RAs, during the 1990s, and the emergence of *Social Sciences* and *Humanities* during the 2000s). Each circle in the network represents a RA. The labels of each RA on the left network correspond to the circles of the networks on the right. The thickness of the lines in the networks is proportional to the number of records that involves two RAs that are linked by the line. The full list of RAs is shown in Table S3.3. The set of network plots for all years is presented in the Supplementary video.

Starting in the late 1980s, and accelerating during the 1990s, interdisciplinarity in invasion research expanded to include ecosystem restoration and management, in so doing incorporating limited social insights. The beginnings are exemplified by contributions from the SCOPE program and the first conference of the *Society for Ecological Restoration* in 1989 (Hobbs and Richardson, 2011). A typical example from our literature search discusses the spread, management, and governance of non-native species (Groves and Burdon, 1986). However, the number of disciplines involved in this apparent growth phase of interdisciplinarity during the 1980s and 1990s remained relatively stable. The interdisciplinary publications of that period tended to include only disciplines closely related to ecology and environmental sciences, namely biology, geosciences, and other natural sciences (Figure 3.3; Supplementary video and Figure S3.1). Engineering and technology, as well as social sciences and humanities were represented sporadically in the 1990s. This trend was followed by a consistent presence and steady diversification of research areas since the 2000s (Figure 3.3; Supplementary video and Figure S3.1), during which the time invasion science also seems to have converged towards a broader, social-ecological endeavour.

3.5. THE ADVENT OF SOCIAL AND SOCIAL-ECOLOGICAL PERSPECTIVES

Our review shows that 92.4% of ecological or social interdisciplinary publications (out of 9192) correspond to records classified as purely ecological, 4.4% correspond to purely social records, and 3.2% (293 records) are classified as social-ecological (Figure 3.1). The 1990s and 2000s were characterised by the advent of purely social (1990s) and then coupled social-ecological research (2000s; Figure 3.4).

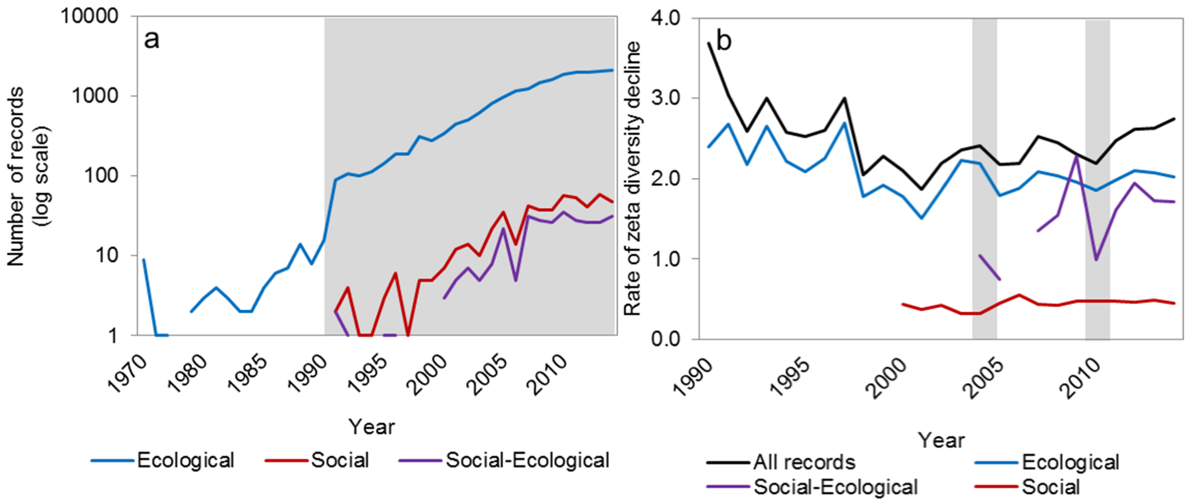

Figure 3.4. The number of records attributed to ecological, social, and social-ecological RAs in a logarithmic scale (a), and the rates of zeta diversity decline calculated considering the whole set of records, and only ecological, social, or social-ecological records (b). Low rates of zeta diversity decline indicate high interdisciplinarity, expressing a higher number of RAs shared by many records, and a fewer number of RAs exclusive to selected records, especially those with fewer RAs. Due to the low number of records and RAs observed during 1950-1990, zeta diversity decline was computed for 1950-1990 as whole, and then for each subsequent year separately. Time periods discussed in detail in the text are shown in light grey.

The slow uptake of the human dimension in invasion research until the 2000s might indicate that, until then, there was a belief that problems associated with invasive species could be solved through technological solutions, building mostly on knowledge about the ecology of invasive species (McNeely, 2001; Simberloff, 2001). Our results show that there was a growing interest in complex mathematical models for elucidating aspects of invasion dynamics during this period (e.g. species distribution models; Guisan and Zimmermann, 2000; Thuiller et al., 2005; Figure 3.3; Supplementary video and Figure S3.1). Yet, despite available technological solutions, management interventions were often considered as unsuccessful, possibly due to the lack of an explicit recognition of the role of the human dimension (e.g. McNeely, 2001; Simberloff, 2001; Chaffin et al., 2016). A second reason might be that biodiversity conservation was, until the emergence of the ecosystem services concept in the 2000s (*Millennium Ecosystem Assessment*, 2005), virtually independent of human valuation and social insights. In contrast, research on environmental hazards and natural disasters, which were always considered as immediate threats to human life and welfare, gave explicit attention to social dimensions earlier (e.g. Dahlberg et al., 2016). A third reason could be that social-oriented publications were considered for indexing later than those from other disciplines; this may have resulted in a delayed coverage by ISI (Leydesdorff and Rafols, 2009; Rafols et al., 2010).

The human dimension of invasions gained wider attention after 2000 (Figures 3.4 and 3.5). One reason was the *Global Invasive Species Programme* (GISP) that fostered close interactions among ecologists, economists, social scientists, and especially policy makers (Kueffer and Hirsch Hadorn, 2008; Davis, 2011; Hui and Richardson, 2017). Another reason might be the emergence of the ecosystem services concept, which provided new research directions relating to invasions - as expressed in our retrieved publications such as Zavaleta (2000), Van Wilgen et al. (2008), and Simberloff et al. (2013).

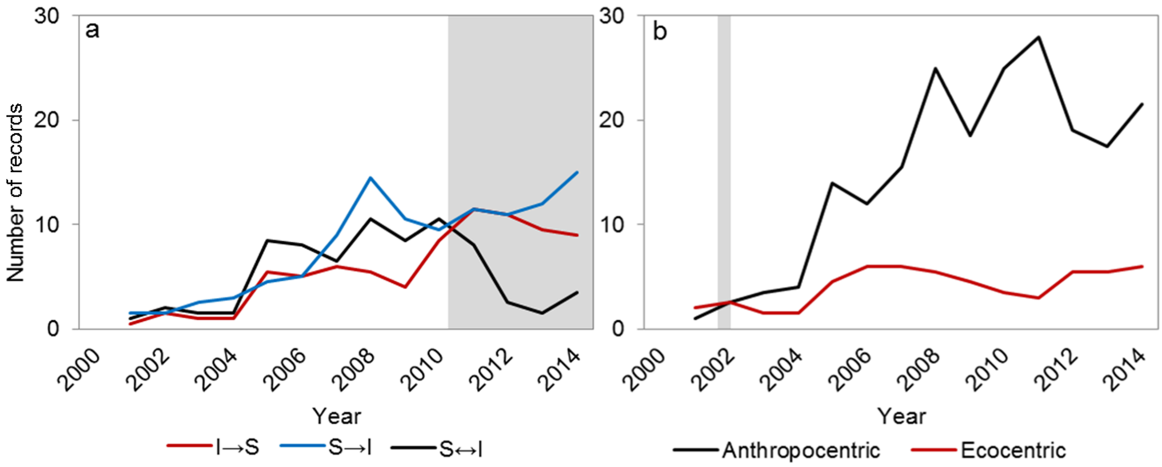

Figure 3.5. The number of social-ecological records for each year (smoothing curves showing averages for 2-year periods), attributed to a specific category regarding: the direction of influence between the social system and the invasion process (a), and the main direction of impacts provoked by the invasion process (b; see Table 3.1 for further explanations). Time periods discussed in detail along the text are shown in light grey.

Social science and humanities perspectives are apparent in publications on the history of the invasion field (e.g. Davis et al., 2001; Davis, 2006) and on the metaphors it mobilises (e.g. Larson, 2005, 2007), and in reports such as *The Great Reshuffling* (McNeely, 2001), a special journal issue on Australian Acacias (Richardson et al. 2011), or the book *Fifty years of invasion ecology* (Richardson, 2011b). The maturation of such perspectives is also reflected in the emergence of stand-alone collections of social science or humanities publications on the topic (e.g. Rotherham and Lambert, 2011; Frawley and McCalman, 2014). The recent interest of social scientists in invasions appears to be largely focused on three subjects: (1) the role of the human influence on the invasion process (McNeely, 2001; Rotherham and Lambert, 2011; Humair et al., 2015) with 44.0% (out of 283) of the records from our dataset conceptualising human activities as drivers of the invasion process (Figure 3.5a); (2) direct or indirect impacts of species establishment on humans (Simberloff et al., 2013; Schindler et al., 2015), with

75.0% (out of 283) of the records (Anthropocentric; Figure 3.5b); and (3) practical aspects of management (Matzek et al., 2013; Head and Atchison, 2015; Figure 3.6 and Table S3.4).

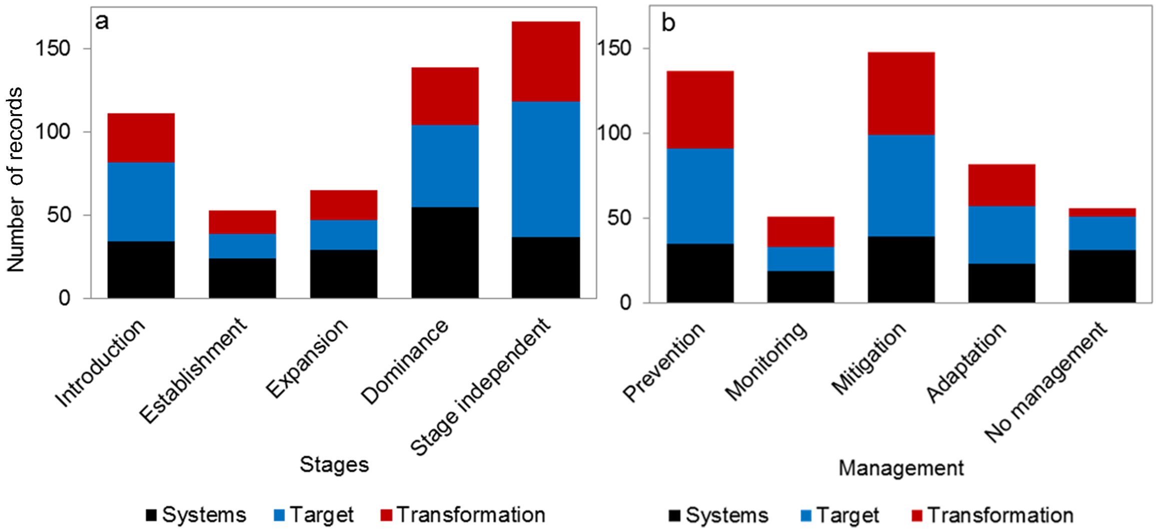

Figure 3.6. The number of social-ecological records for the time-period of 2000-2014, attributed to a specific stage of the invasion process (a), and type of management strategy addressed (b; see Table 3.1.1 for further explanations). The figure also shows the distribution of knowledge dimensions (systems, target, and transformation knowledge) across the invasion stages and management strategies.

Nonetheless, contributions from the social sciences and humanities still comprise a minor proportion of the invasion literature, making up less than 5.0% of the canon (Figures 3.4, 3.5; see also Figure S3.1). This is likely because the focal topic (biological invasions) was framed, defined, and elaborated foremost as an ecological phenomenon, and most of the key questions that feature prominently in research agendas still draw most interest from ecologists. The volume of basic and applied ecological, environmental, and management publications on invasions (with 75.1% of the 11465 interdisciplinary publications on invasions, corresponding to 8496 records; Figure 3.1) is unsurprisingly larger than that of social science or the humanities on these themes; work in the social realm has largely emerged in reaction to ecological ideas and management actions. Most social-oriented research relating to invasions has critiqued management activities, or has addressed the philosophical, ethical, or conceptual underpinnings of the field (Carruthers et al., 2011; Estevez et al., 2014; Frawley and McCalman, 2014).

However, our results must be interpreted with caution, as differences in publication culture between ecological/environmental sciences and social science/humanities may have limited the representation of the latter in the literature covered by ISI. Social sciences and the humanities show different citation behaviour, publish more in books and journals that may not be catalogued as comprehensively by ISI, thus potentially resulting in an underrepresentation of contributions in our treatment (Leydesdorff and Rafols, 2009; Rafols et al., 2010).

3.6. THE CURRENT DEFICIT OF SOCIAL-ECOLOGICAL APPROACHES IN INVASION RESEARCH

Biological invasions are increasingly recognised as a social-ecological phenomenon (McNeely, 2001; Kueffer and Hirsch Hadorn, 2008; Kueffer, 2013; Estevez et al., 2014; Head et al., 2015; Hui and Richardson, 2017). However, our literature survey shows that neither social-ecological research, nor explicit interdisciplinarity and integration of feedbacks between the social and the ecological systems are easily found in invasion studies (Figures 3.3-3.5). The 283 social-ecological studies found are relatively equally distributed across the different invasion stages, management strategies, and knowledge dimensions (Figure 3.1.6). For all invasion stages and management strategies, studies analyse the drivers of invasions (systems knowledge; corresponding to 32.0% of social-ecological records), their valuation (target knowledge; 28.0%) and solutions to target them (transformation knowledge; 40.0%) from a social-ecological perspective.

Specifically, the set of social-ecological studies on systems knowledge which we found are focused on how humans shape the context for invasion, and thereby facilitate or hinder aspects of the invasion process. Specific papers from our search explore, for instance, how diverse social factors (such as government programs, people's beliefs, and socioeconomic status) relate with the conversion of non-invaded to invaded habitats (Brenner, 2010), and how the social system affects invasion processes at different levels, through e.g. ineffective control of immigration borders or illegal trade (Rodríguez-Labajos et al., 2009). Likewise, social-ecological studies focused on valuation (target knowledge), examine how people perceive invasive species, highlighting the need to account for cultural influences and normative issues (Rotherham and Lambert, 2011; Tassin and Kull, 2015; Essl et al., 2017; Kueffer and Kull, 2017). Examples from our search include xenophobic standpoints regarding the cohabitation with non-native species (Larson, 2005; Estevez et al., 2014), or aspects of valuation implicit in metaphors used in scientific writing (Larson, 2005, 2013; Kueffer and Larson, 2014). These examples comprise research on people's thoughts, emotions, and representations, as well as cultural and knowledge differences regarding meanings and intentions towards invasive species (Larson, 2005; Hall, 2009; Buijs et al., 2012; Heger et al., 2013). Finally, social-ecological studies focused on more effective management solutions (transformation knowledge) include integrative solutions regarding conflicts of interest, work capacity, efficiency, and legitimacy of individuals and groups that manage (or use) invasive species or invaded areas, as well as their articulation with social institutions, frameworks, and rules (Kull et al., 2011; Matzek et al., 2013; Simberloff et al., 2013; Estevez et al., 2014; Essl et al., 2017). Examples from our search include the evaluation of enforcement and inspection regimes in firms for reducing invasion risk, both in terms of resource allocation and effectiveness of policies (Ameden et al., 2009); participatory processes with stakeholders such as the horticulture industry (Humair et al., 2014b); and approaches focused on how public advertising can increase society outreach and influence behaviour towards managing invasions (Shaw et al., 2014).

3.7. Bringing social-ecological approaches to the centre of invasion research

Despite progress, achieving interdisciplinarity seems to still constitute a challenge to invasion science. To reduce the ecological-environmental focus of invasion science and pave the way for higher cross-fertilisation with the social sciences and humanities, we suggest that framing problems, methods, and applications in invasion research needs to be rethought (also following Larson, 2007; Kueffer and Hirsch Hadorn, 2008; Hattingh, 2011). The examples discussed above provide a range of entry points for initiating the reframing research questions in invasion science as a social-ecological challenge, with the aim of overcoming the rooting of the field in a purely ecological perspective. Further entry points are necessary to help unlock the potential for more interdisciplinary, social-ecological thinking (also Liu et al., 2007; Hui and Richardson, 2017). The starting point for research might then not simply be "*the introduction/invasion of species X in ecosystem Y*", but instead the "*interlinked social-ecological changes in region Z*". This would still permit focused ecological research on X and Y, but would also pave the way for broader perspectives and invite interdisciplinary collaboration (and publication) from the perspective of (and with collaborators from) the social sciences and humanities (Larson, 2007, 2011). This could also overcome the monodisciplinary nature of invasion science and allow a more genuine integration of disparate disciplines, each of which would bring their own key issues and research cultures and identify joint research questions and linking methods (after Kueffer and Hirsch Hadorn, 2008; Rissman and Gillon, 2016). It would be beneficial, for instance, to promote debates that target the social construction of invasive species based on scientific facts or cultural norms (Hattingh, 2011;

Larson, 2011; Estevez et al., 2014; Tassin and Kull, 2015; Kueffer and Kull, 2017), and to welcome stakeholders besides academics. Thus, practitioners, scholars from ecology, and social scientists could be called upon not only to address pre-defined topics arising from ecological studies or resource management challenges (and vice versa; Davis, 2011; Tengo et al., 2014; N'Guyen et al., 2016), but also to shape new ideas, concepts, and research questions, and to apply new approaches and methodologies for addressing these questions and to participate in communicating results to multiple stakeholders (Kueffer and Hirsch Hadorn, 2008; Hattingh, 2011; Richardson et al., 2011; Heger et al., 2013; Courchamp et al., 2017).

Repackaging invasion science as a field explicitly oriented toward a variety of questions centred on interlinked social-ecological dynamics would open more opportunities for merging insights from science, policy management and society to understand, deliberate, mitigate, manage, and adapt to biological invasions (Courchamp et al., 2017). Such a reframing could build on recent work on invasion management (Head and Atchison, 2015; N'Guyen et al., 2016; Woodford et al., 2016) on the social, political, and economic context (Carruthers et al., 2011; Kull et al., 2011), and on the communication with the broader public (Kueffer and Hirsch Hadorn, 2008; Heger et al., 2013; Estevez et al., 2014; Kueffer and Larson, 2014; Tassin and Kull, 2015; Courchamp et al., 2017). Lastly, invasion science could benefit from the recent developments in social-ecological systems theory or resilience thinking (Liu et al., 2007; Cote and Nightingale, 2012; Frawley and McCalman, 2014). These promising approaches include a human perspective on invasions, which goes beyond the unsatisfactory "threat to native species" or "good versus bad" dichotomy (Larson, 2007), and thus pave the way for "*governing invasive species in a more integrated and cost-efficient manner given a renewed focus on understanding and managing ecosystem dynamics as opposed to single species*" (Chaffin et al., 2016, p. 405).

The integration of real human-environment interactions could provide important opportunities for deliberating and forging solutions based on multiple (actor) interests and uncertainties, not only when dealing with invasions (Kueffer and Hirsch Hadorn, 2008; Kull et al., 2011; Matzek et al., 2013; Head et al., 2015), but also with other social-ecological phenomena (see e.g. Tengo et al., 2014; Bennett et al., 2017). Framing invasions from a more balanced social-ecological perspective would help to, among other things, clarify distinct viewpoints relating to perceptions of risks and opportunities, and would help in decision-making by applying collaborative and participatory approaches that could not be achieved through traditional approaches (Kueffer and Hirsch Hadorn, 2008; Heger et al., 2013; Kueffer, 2013; Estevez et al., 2014; Tassin and Kull, 2015; Chaffin et al., 2016; Courchamp et al., 2017).

3.8. Conclusions

We documented the growth of invasion science that has been rooted in ecology and has targeted an environmental problem. We provided explicit quantitative data on the invasion literature since 1950. Although interdisciplinarity has become more prominent as the field has grown, collaborations between disciplines remain largely confined within subdisciplines of ecology and the environmental sciences. The social sciences and humanities have taken an increasing interest in invasions in the last decade, but collaborations between social scientists and ecologists, and truly integrative social-ecological studies remain difficult to capture in quantitative literature searches. This is despite many calls for such studies given the social-ecological nature of invasions, their valuation, and management (following Larson, 2007; Kueffer and Hirsch Hadorn, 2008; Kueffer, 2013; Head et al., 2015; Tassin and Kull, 2015; Chaffin et al., 2016, among others). The distinct culture of social sciences and humanities concerning publication and citation approaches could have influenced the limited illustration of social and social-ecological records in our search. Nevertheless, the few social-ecological studies that we found indicate the high potential for diverse social-ecological research to address the increasingly complex dimensions of invasion science and management.

Invasion science has been punctuated by several key events over its short history. We suggest that the time is ripe for invasion science to adjust its course to the following: (1) form research teams comprising a balanced pool of social scientists (including scholars from the humanities) and ecologists (and other natural scientists) with common strategies for science disclosure; (2) establish long-term and reciprocal relationships with multiple stakeholders addressing conceptual questions, research problems, and collaborative management approaches (also N'Guyen et al., 2016); (3) encourage workshops and other forms of interaction to design novel and integrative conceptual frameworks that explicitly challenge and extend existing frameworks, methodologies, theories, and problem-framings in invasion science (also Heger et al., 2013); and (4) create arenas for social-ecological systems thinking that move beyond the classical dichotomy of invasions as beneficial (ecosystem service providers) or harmful (drivers of ecosystem disservices) to society (also Larson 2007; Vaz et al., 2017).

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SUPPLEMENTARY MATERIAL II

The supplementary video associated to this research can be found here: https://link.springer.com/article/10.1007/s13280-017-0897-7#SupplementaryMaterial

Appendix A - Details on the literature search

Table S3.1. Search performed in ISI Web of Science (from February to September 2015). A set of search terms was compiled based on a list derived from a number of core references (see below) and the expert knowledge of the research team. The research team included members with a disciplinary as well as interdisciplinary background and experience in interdisciplinary collaboration, i.e., from biology, ecology, human geography, and mathematics. The final search string (in bold in table below) was derived through an iterative procedure, by (1) reviewing a short list of key publications, and including pertinent keywords for the search, (2) checking the records retrieved by the search, and including or excluding pre-existent and new keywords, and (3) re-conducting the search with the new set of keywords. New terms were step-by-step added and then the first 10 hits were checked for relevance. If the new search results were an improvement over the old ones the new term was kept, else it was removed. The search was performed in the field 'topic' of ISI Web of Science for records from 1950-2014. We considered records written in English, French, Spanish or Portuguese.

Keywords on TOPIC	**Hits**	**Comments**
"Ecological invasion*"	63	*Add new term*
"Ecological invasion*" OR "Biological invasion*"	5739	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology"	6042	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology"	6220	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species"	14813	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species"	16667	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species" OR "Introduced species"	19050	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species" OR "Introduced species" OR "Non-native species"	19971	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species" OR "Introduced species" OR "Non-native species" OR "Nonnative species"	20175	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species" OR "Introduced species" OR "Non-native species" OR "Nonnative species" OR "Nonindigenous species"	20333	*Add new term*

"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species" OR "Introduced species" OR "Non-native species" OR "Nonnative species" OR "Nonindigenous species" OR "Non-indigenous species"	20646	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species" OR "Introduced species" OR "Non-native species" OR "Nonnative species" OR "Nonindigenous species" OR "Non-indigenous species" OR "allochthonous species"	20695	*Add new term*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species" OR "Introduced species" OR "Non-native species" OR "Nonnative species" OR "Nonindigenous species" OR "Non-indigenous species" OR "allochthonous species" OR "Exotic species"	**23640**	***Add new term***
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species" OR "Introduced species" OR "Non-native species" OR "Nonnative species" OR "Nonindigenous species" OR "Non-indigenous species" OR "allochthonous species" OR "Exotic species" OR "Released species"	23711	*Drop "released species" due to irrelevant hits*
"Ecological invasion*" OR "Biological invasion*" OR "Invasion biology" OR "Invasion ecology" OR "Invasive species" OR "Alien species" OR "Introduced species" OR "Non-native species" OR "Nonnative species" OR "Nonindigenous species" OR "Non-indigenous species" OR "allochthonous species" OR "Exotic species" OR Invader	27122	*Drop "invader" due to irrelevant hits*

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Appendix B - Inclusion and exclusion criteria applied to the main dataset.

Table S3.2. The records retrieved by the search in Web of Science (total number, n = 23640) were subjected to inclusion and exclusion criteria to eliminate non-relevant information for the research goals. These criteria related to both the type of record, and the population being targeted by the record retrieved by the search. The criteria were applied by checking the title and keyword of each record. We considered records written in English, French, Spanish or Portuguese (where the English keywords were at least referred in the title of the record). After applying these criteria, the final dataset consisted of 23390 records.

Criteria type	**Exclusion criteria**	**Inclusion criteria**
Record type	Biographical items, corrections/corrigendum, items about an individual, poetry, and anonymous documents which could not be found for checking veracity.	Research articles, book chapters, book reviews, editorial material, letters, meeting abstracts, news items, notes, proceeding papers, reviews.
Targeted population	Records which focus on alien species in relation to space (mostly in astronomy, astrophysics and physics); alien species linked to literature and films with no connection to the real world; human population considered alien species (e.g., woman as an alien species; mostly from literature and movies); invasive species from a poetic perspective and attributed to human population (mostly religious and philosophical studies); clinical terms which use alien species for referring to an organism outside the human body (mostly in dentistry, ophthalmology, dermatology, oncology) or animals in laboratory experiences (clinical laboratory).	All records unless otherwise stated in the exclusion criteria.

Appendix C - List of ISI research areas.

Table S3.3. List of ISI research areas attributed to the records retrieved by our search, with the classification into research fields. Articles were classified according to nine broad research fields (RF) based on 110 subordinate research areas (RA). RAs were slight adaptations of those used by ISI Web of Science (ISI WOS). Classification into RFs was conducted by an interdisciplinarity team (with academics from e.g., mathematics, biology, ecology, and social sciences), and followed previous work by Leydesdorff and Rafols (2009), Porter and Rafols (2009), Rafols et al. (2010), and Wagner et al. (2011). This classification was adapted to best represent the most intuitive combinations of RAs in the biological invasion literature, and to facilitate the dissemination of the set of RAs retrieved by our search (reason why the fields of ecology, environment, biology, and (other) natural sciences were considered as separated research fields; whereas, the broad research fields of social sciences and humanities were not subdivided).

Original names of RAs adopted by ISI WOS	Names of the research areas (RAs) adopted in the study	Classification of RAs into research fields (RFs)*
Acoustics	Acoustics	Engineering & Technology
Agriculture	Agriculture	Environmental Sciences
Allergy	Allergy	Medicine & Health Sciences
Anatomy & Morphology	Anatomy	Medicine & Health Sciences
Anthropology	Anthropology	Humanities
Archaeology	Archaeology	Humanities
Architecture	Architecture	Humanities
Area Studies	Area Studies	Social Sciences
Art	Art	Humanities
Arts & Humanities - Other Topics	Humanities	Humanities
Audiology & Speech-Language Pathology	Audiology	Medicine & Health Sciences
Automation & Control Systems	Automation	Engineering & Technology
Behavioral Sciences	Behaviour	Biology
Biochemistry & Molecular Biology	Biochemistry	Biology
Biodiversity & Conservation	Biodiversity	Ecology & Evolution
Biophysics	Biophysics	Biology
Biotechnology & Applied Microbiology	Biotechnology	Biology
Business & Economics	Economics	Social Sciences
Cell Biology	Cell	Biology
Chemistry	Chemistry	(Other) Natural Sciences
Communication	Communication	Social Sciences
Computer Science	Computer	Engineering & Technology
Construction & Building Technology	Construction	Engineering & Technology
Cultural Studies	Culture	Humanities
Demography	Demography	Social Sciences

Developmental Biology	Development	Biology
Education & Educational Research	Education	Social Sciences
Electrochemistry	Electrochemistry	(Other) Natural Sciences
Emergency Medicine	Emergency	Medicine & Health Sciences
Endocrinology & Metabolism	Endocrinology	Medicine & Health Sciences
Energy & Fuels	Energy	Engineering & Technology
Engineering	Engineering	Engineering & Technology
Entomology	Entomology	Ecology & Evolution
Environmental Sciences & Ecology	Environment	Environmental Sciences
Ethnic Studies	Ethnic	Social Sciences
Evolutionary Biology	Evolution	Ecology & Evolution
Fisheries	Fisheries	Environmental Sciences
Food Science & Technology	Food	Engineering & Technology
Forestry	Forestry	Environmental Sciences
General & Internal Medicine	Internal Medicine	Medicine & Health Sciences
Genetics & Heredity	Genetics	Biology
Geochemistry & Geophysics	Geochemistry	Geosciences
Geology	Geology	Geosciences
Government & Law	Law	Humanities
Health Care Sciences & Services	Health Care	Medicine & Health Sciences
History	History	Humanities
History & Philosophy of Science	History & Philosophy	Humanities
Imaging Science & Photographic Technology	Imaging	Engineering & Technology
Immunology	Immunology	Medicine & Health Sciences
Infectious Diseases	Diseases	Medicine & Health Sciences
Information Science & Library Science	Information	Social Sciences
Instruments & Instrumentation	Instrumentation	Engineering & Technology
Integrative & Complementary Medicine	Integrative Medicine	Medicine & Health Sciences
International Relations	Relations	Social Sciences
Legal Medicine	Legal Medicine	Medicine & Health Sciences
Life Sciences & Biomedicine - Other Topics	Biomedicine	Medicine & Health Sciences
Linguistics	Linguistics	Social Sciences
Literature	Literature	Humanities
Marine & Freshwater Sciences	Marine	Ecology & Evolution

Materials Science	Materials	Engineering & Technology
Mathematical & Computational Biology	Mathematics Biology	Biology
Mathematical Methods In Social Sciences	Mathematics Social	Social Sciences
Mathematics	Mathematics	(Other) Natural Sciences
Mechanics	Mechanics	Engineering & Technology
Meteorology & Atmospheric Sciences	Meteorology	Geosciences
Microbiology	Microbiology	Biology
Microscopy	Microscopy	Engineering & Technology
Mining & Mineral Processing	Mining	Engineering & Technology
Mycology	Mycology	Ecology & Evolution
Neurosciences & Neurology	Neurology	Medicine & Health Sciences
Nuclear Science & Technology	Nuclear Science	Engineering & Technology
Nutrition & Dietetics	Nutrition	Medicine & Health Sciences
Obstetrics & Gynecology	Obstetrics	Medicine & Health Sciences
Oceanography	Oceanography	Geosciences
Operations Research & Management Science	Operations	Social Sciences
Optics	Optics	Engineering & Technology
Paleontology	Paleontology	Biology
Parasitology	Parasitology	Biology
Pathology	Pathology	Medicine & Health Sciences
Pharmacology & Pharmacy	Pharmacology	Medicine & Health Sciences
Philosophy	Philosophy	Humanities
Physical Geography	Physical Geography	Geosciences
Physics	Physics	(Other) Natural Sciences
Physiology	Physiology	Biology
Plant Sciences	Plant Sciences	Ecology & Evolution
Psychology	Psychology	Social Sciences
Public Administration	Administration	Social Sciences
Public, Environmental & Occupational Health	Public Health	Medicine & Health Sciences
Radiology, Nuclear Medicine & Medical Imaging	Radiology	Medicine & Health Sciences
Religion	Religion	Humanities
Remote Sensing	Remote Sensing	Engineering & Technology
Reproductive Biology	Reproduction	Medicine & Health Sciences
Research & Experimental Medicine	Research Medicine	Medicine & Health Sciences
Robotics	Robotics	Engineering & Technology

Science & Technology - Other Topics	Technology	Engineering & Technology
(Social) Geography	Geography	Social Sciences
Social Issues	Social Issues	Social Sciences
Social Sciences - Other Topics	Social Sciences	Social Sciences
Sociology	Sociology	Social Sciences
Spectroscopy	Spectroscopy	Engineering & Technology
Sport Sciences	Sport	Medicine & Health Sciences
Telecommunications	Telecommunications	Engineering & Technology
Toxicology	Toxicology	Environmental Sciences
Transportation	Transportation	Engineering & Technology
Tropical Medicine	Tropical Medicine	Medicine & Health Sciences
Urban Studies	Urban	Environmental Sciences
Veterinary Sciences	Veterinary	Medicine & Health Sciences
Virology	Virology	Medicine & Health Sciences
Water Resources	Water	Environmental Sciences
Zoology	Zoology	Ecology & Evolution

***Description of research fields:**

Biology - Includes biological RAs that are not covered by any of the other RAs (Ecology & Evolution and Environmental Sciences). These include for instance theoretical biology, mathematical biology, biophysics, cell biology.

Ecology & Evolution - Include RAs concerning many areas relating to the study of the interrelationship of organisms and their environments, including evolutionary ecology, biogeography, marine ecology, wildlife and biodiversity research.

Engineering & Technology - Include RAs concerning the application of engineering and technology to deal with a wide range of issues, including the construction, design and operation of equipment and tools used for different purposes.

Environmental Sciences - Include the study of the environment, its contamination, toxicology, health, monitoring and management, including soil science and conservation, water resources research and engineering, and climate change.

Geosciences - Include RAs with a general approach to the study of the Earth, including geology, geochemistry/geophysics, meteorology and atmospheric sciences.

Humanities - Comprise the RAs with a focus on human culture, including arts, religion, history, and anthropology.

Medicine & Health Sciences - Covers medical research related to human and animal health, including public health and sports studies.

(Other) Natural Sciences - Include those RAs focused on the description and understanding of natural phenomena, and that could not be accounted in other categories, including mathematics, chemistry and physics.

Social Sciences - Include RAs with a focus on society and their intrinsic relationships, including a variety of RAs which range from economics to public management and sociology.

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Appendix D - Focal questions and categories considered.

Table S3.4. Focal questions and categories considered, and classification of records (n = 283) resulting from the literature review process. This review process was considered in the final step of our analytical framework (Step 4), in which we cross-checked the set of records classified as social-ecological (n = 293 out of 23393). Each individual record was reviewed to confirm its social-ecological scope, by subsequently screening the title, keywords and abstract of each record. After removing unsuitable articles, the complete text of the final set of records (n = 283) was reviewed analysed to answer a set of focal questions related to our research goals.

Codes	Description	% records
STEP 4A. Social-ecological approaches		
A1. What is the main direction of influence between the social system and the invasion process?		
Social→Invasion (S→I)	The social system drives the invasion process	**44**
Invasion→Social (I→S)	The invasion process influences the social system	**29**
Invasion↔Social (S↔I)	There is reciprocity between the two systems	**27**
A2. What is the main direction of impacts provoked by the invasion process?		
Anthropocentric	The invasion process provokes impacts (partially/totally) on the social system	**75**
Ecocentric	The invasion process provokes impacts exclusively on the ecosystem	**25**
A3. Which form of knowledge dimension does the study produce?		
Systems knowledge (‘causes’)	Oriented towards analysing and improving the causal understanding of the invasion process	**32**
Target knowledge (‘valuation’)	Oriented towards clarifying conflicts of interests and values, including peoples’ perceptions, valuations and conceptualizations	**28**
Transformation knowledge (‘solutions’)	Oriented towards improving or avoiding a particular situation related to the invasion process	**40**
STEP 4B. Invasion process		
B Which component of the invasion process is studied?		
Introduction	Focuses on the pathways of species introduction from one geographical region to another	**18**
Establishment	Focuses on the determinants of success of species establishment	**10**
Expansion	Focuses on the patterns and mechanisms of species expansion	**12**
Dominance	Focuses on patterns and processes related to invaders that have become dominant in an invaded area, including impacts and management	**27**
Stage independent	Studies that do not specify the stage of the invasion process, mostly because they address the invasion process as a whole	**33**
STEP 4C. Management type		
C. Which aspects of invasion management are considered by the study?		
Prevention	Focuses on preventing the introduction of new invasive species (including risk assessments of source areas, transportation pathways, and species characteristics)	**26**
Monitoring	Focuses on mapping, assessing and monitoring the distribution and impacts of invasive species	**11**

Mitigation	Focuses on reducing the (likelihood) of impacts of an invasive species, including containment of further spread and eradication	**30**
Adaptation	Focuses on dealing with, and tolerating impacts (including tolerating species or using them, e.g., for timber, medicinal or ornamental goals)	**15**
No management	There is no focus on the management of invasive species	**12**
Unspecified	The study does not specify the management type	**6**

Appendix E - Number of observations classified into one of the main research fields

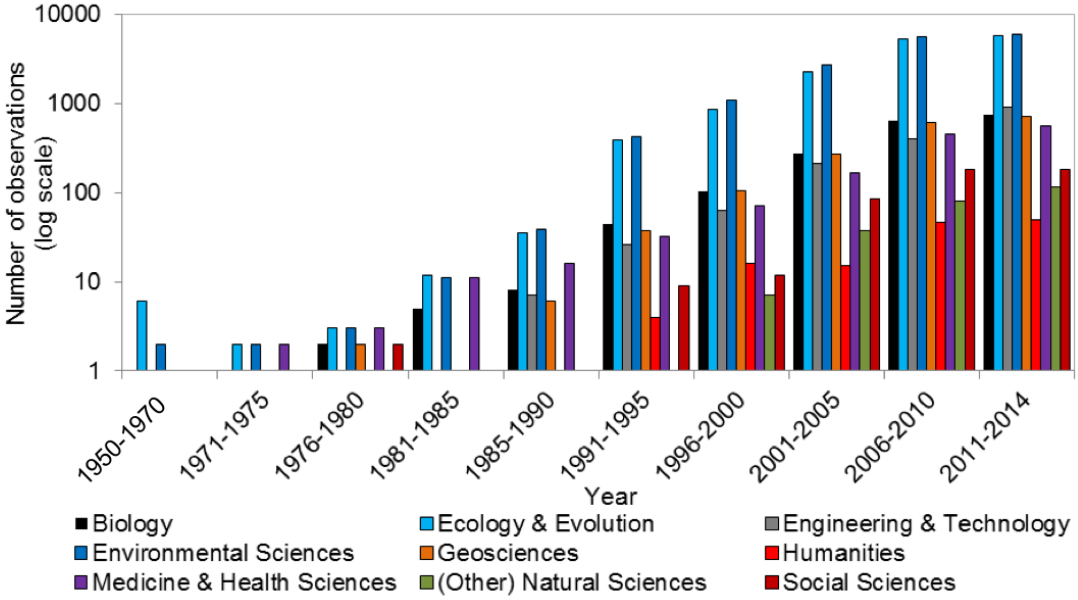

Figure S3.1. Bar plot showing the number of records/observations classified into one of the main research fields. Values in the y-axis are expressed in a logarithmic scale.

CHAPTER 4. SOCIAL-ECOLOGICAL ANALYSIS OF CULTURAL ECOSYSTEM SERVICES

DISCLAIMER

This chapter is an original contribution of this thesis, published as an original paper in journal *Ecological Indicators* under the title "*An indicator-based approach to analyse the effects of non-native tree species on multiple cultural ecosystem services*". The paper had the contribution of the following authors:

Ana Sofia Vaz[a,b,*], Pilar Castro-Díez[c], Oscar Godoy[d], Álvaro Alonso[c], Montserrat Vilà[e], Asunción Saldaña[c], Hélia Marchante[f,g], Álvaro Bayón[e], Joaquim S. Silva[a,g,h], Joana R. Vicente[a,i], João P. Honrado[a,b]

[a]Research Network in Biodiversity and Evolutionary Biology, Research Centre in Biodiversity and Genetic Resources (InBIO-CIBIO), Universidade do Porto, Campus Agrário de Vairão, Rua Padre Armando Quintas, PT4485-661 Vairão, Portugal;

[b]Faculdade de Ciências, Universidade do Porto, Rua do Campo Alegre, s/n, PT4169-007 Porto, Portugal;

[c]Departamento de Ciencias de la Vida, Unidad Docente de Ecología, Facultad de Biología, Ciencias Ambientales y Química, Universidad de Alcalá, E-28805, Alcalá de Henares, Spain;

[d]Instituto de Recursos Naturales y Agrobiología de Sevilla (IRNAS-CSIC), E-41080 Sevilla, Spain;

[e]Department of Integrative Ecology, Estación Biológica de Doñana (EBD-CSIC), Avda. Américo Vespucio 26, Isla de la Cartuja, E-41092 Sevilla, Spain;

[f]Centre for Functional Ecology, Department of Life Sciences, University of Coimbra, Calçada Martim de Freitas, 3000-456, Coimbra, Portugal;

[g]Escola Superior Agrária, Instituto Politécnico de Coimbra, Bencanta, 3045-601, Coimbra, Portugal;

[h]Centre for Applied Ecology "Prof. Baeta Neves" (InBIO-CEABN), Instituto Superior de Agronomia, Universidade de Lisboa, Tapada da Ajuda, PT1349-017 Lisboa, Portugal;

[i]Laboratory of Applied Ecology, CITAB - Centre for the Research and Technology of Agro-Environment and Biological Sciences, University of Trás-os-Montes e Alto Douro, Vila Real, Portugal.

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ABSTRACT

Limitations in the assessment of cultural ecosystem services through quantifiable approaches have constrained our knowledge of how these services can be altered by drivers of global change, such as non-native tree species. Here, we address this caveat by evaluating the effects of non-native tree species, in comparison to native ones, on several categories of cultural services, i.e., recreation and ecotourism, aesthetics, inspiration, and cultural heritage. We propose an indicator-based approach that includes the use of a meta-analysis statistics, the odds ratio, to evaluate photographic, internet and catalogue data, and to infer on the effects of non-native trees on cultural services. We apply our approach to the Iberian Peninsula, exploring potential environmental and socio-economic predictors of non-native tree effects across NUTS-2 administrative regions. Overall, non-native tree effects differed among categories of cultural services and varied with the data type. Non-native trees increased recreation and ecotourism services, when considering data from official tourism entities, but not from nature route users. Data from inventories of urban parks and catalogues of ornamental plant dealers suggest that non-native trees decreased aesthetics services, particularly in Spain, but not in Portugal. Non-native trees also increased cultural heritage services, but no significant effects were observed on inspiration services. Overall, non-native trees showed higher increases in cultural services across regions with lower levels of development (in terms of income, employment and education) and life satisfaction. We suggest that management should emphasise awareness on non-native trees, including the risks involved in promoting the expansion of potentially invasive species. Efforts to raise awareness should prioritise official tourism entities and ornamental plant dealers, with a special focus on less developed regions. Our proposed approach represents a pioneer assessment of the relations between non-native trees and cultural ecosystem services, supporting strategic management in Iberia. The focus on widely available data sources enables reproducibility and application in assessments worldwide.

Keywords: Aesthetics; Alien plants; Cultural heritage; Inspiration; Meta-analysis; Recreation and ecotourism

4.1. Introduction

The growing recognition of nature's contributions to human well-being has fostered research on ecosystem services (Blicharska et al., 2017; MEA 2005; Schröter et al., 2016). Besides provisioning (e.g., drinking water, secure food) and regulating (e.g., hazard mitigation, pollination) services, ecosystems also provide cultural services. The *Millennium Ecosystem Assessment* (MEA 2005; p. 40) defines cultural ecosystem services as the "*nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences*", including inspiration and cultural heritage (see also Chan et al., 2012; Fish et al., 2016).

Cultural ecosystem services are relevant in various governance contexts, such as land tenure and management, recreation revenues, and human identity and traditions (Carruthers et al., 2011; Plieninger and Bieling, 2012). However, difficulties in the assessment of cultural services, arising mostly from their subjectivity and difficult quantification, have hampered their consideration in decision-making (Chan et al., 2012; Fish et al., 2016; Schröter et al., 2016). Examples of emerging approaches to asses cultural services include: the use of historical records and vegetation mapping to obtain quality indices of landscape aesthetics or heritage (e.g. Tengberg et al., 2012); public opinion polls to identify cultural benefits (e.g. Poe et al., 2016); monetary evaluations of ecosystem properties (e.g. van Berkel and Verburg, 2014); and the consideration of ecosystem features *per se* as surrogates of cultural services (such as birds, coloured flowers; e.g. Soliveres et al., 2016). The use of social media, namely photographic and (other) internet information, has been recently suggested as a promising approach (e.g. Oteros-Rozas et al., 2017). Coupled with traditional data sources (namely land cover), social media data can offer novel insights on human-nature relations (Figueroa-Alfaro and Tang, 2017).

Understanding how cultural services may be changed by drivers of global change, such as the occurrence of non-native tree species, is a challenge requiring attention (Hernández-Morcillo et al., 2013; Milcu et al., 2013; Oteros-Rozas et al., 2017). Non-native trees can be defined as tree species that were introduced by humans to new geographic areas (Richardson and Rejmánek, 2011). Non-native trees have been introduced for various purposes aiming to increase ecosystem services, mainly wood production, landscape restoration, and ornamental values (Dickie et al., 2014; Kueffer and Kull, 2017; Kull et al., 2011). They provide key resources worldwide, supporting daily basic needs of local communities and economic revenues in forestry and agro-forestry systems (Kull et al., 2011; Vaz et al., 2017a).

Several environmental factors influence the performance of non-native trees in introduced areas (Brundu and Richardson, 2016; Carruthers et al., 2011). Climate and land cover, among others, shape habitat conditions that may constrain or promote the occurrence and performance of non-native trees (Richardson et al., 2014; van Wilgen et al., 2011; Vicente et al., 2016), and thus their effects on ecosystem services. For example, the aesthetic value of non-native trees is influenced by their occurrence, abundance and physiology (Kueffer and Kull 2017), which are inevitably determined by environmental conditions (Richardson et al., 2014; Vicente et al., 2016).

Non-native trees can also decrease ecosystem services and even promote ecosystem disservices, especially when spreading outside plantations, becoming invasive and competing with service-provider native species (Brundu and Richardson, 2016; Pyšek et al., 2012; Krumm and Vítková 2016; Vilà and Hulme, 2017). Many studies already highlighted that non-native species can reduce provisioning and regulating services, such as water provision and soil stabilisation (e.g., Castro-Díez et al., 2014a; Carruthers et al., 2011; Dickie et al., 2014; Pyšek et al., 2012). However, compared to other types of ecosystem services, their effects on cultural services have seldom been investigated (Kueffer and Kull, 2017; Vilà and Hulme, 2017).

It has been suggested that the cultural value of non-native trees depends on visual attributes, such as landscape monotony and homogenisation (e.g. large plantations or invaded areas) or “out-of-normal” and “exotic” features (e.g. large leaves, colourful flowers; Kueffer and Kull, 2017). Non-native trees can also be valued as historical or scientific assets (e.g. from overseas expeditions; Carruthers et al., 2011; Crews, 2003). Most research so far has focused on narratives related to heritage, folklore and tradition (e.g. Carruthers et al., 2011; Kueffer and Kull, 2017; Kull et al., 2011). Examples include the use of non-native species as monumental trees in Italy (Asciuto et al., 2015); the adoption of *Eucalyptus* species in South Africa, *Pinus* species in New Zealand, and *Rhamnus* and *Salix* species in Australia for leisure activities (Dickie et al., 2014); or the use of *Acacia* species in South Africa for cultural ceremonies (Kull et al., 2011).

The cultural value of non-native trees may depend on socio-economic (e.g. education, market values) and welfare factors that influence human perceptions, judgements and attitudes towards these species (Brundu and Richardson, 2016; Krumm and Vítková, 2016). For instance, wealthy countries are more likely to foster the trade and maintenance of non-natives (also Humair et al., 2015; Vilà and Pujadas, 2001), and thus their effects on cultural services.

Education and awareness also influence the way non-native species and respective cultural services are perceived by people (Carruthers et al., 2011; Kueffer and Kull, 2017). Understanding the relations between non-native trees and cultural services across relevant environmental and socio-economic factors could contribute to better management (Dickie et al., 2014; Vaz et al., 2017a). Specifically, it could help in deliberating risks and opportunities associated to non-native trees (Carruthers et al., 2011; Kueffer and Kull, 2017), while converging with sustainability goals and human well-being (Ghosh and Traverse, 2005; Vaz et al., 2017b).

The Iberian Peninsula (Portugal and Spain) has been the target of many introductions of non-native tree species. Some of these species are restricted to urban areas as ornamentals e.g., *Jacaranda mimosifolia* D.Don. However, many others, such as *Ailanthus altissima* (Mill.) Swingle (tree of heaven), *Eucalyptus globulus* Labill. (tasmanian blue gum), *Acacia longifolia* (Andrews) Willd. (long-leaved wattle), *Pinus radiata* D. Don (monterey pine), *Pseudotsuga menziesii* (Mirb.) Franco (douglas fir), *Quercus rubra* L. (red oak) and *Robinia pseudoacacia* L. (black locust), have become widespread (e.g., Castro-Díez et al., 2014a; Sanz Elorza et al., 2004; Vicente et al., 2016). Concern on non-native tree species (either planted, naturalised or invasive) is growing, as they can compete with native biodiversity and change provisioning and regulating services (e.g. related to soil regulation and water provision; Castro-Díez et al. 2014b; Godoy et al., 2010; Morais et al., 2017; Vicente et al., 2016). However, to our knowledge, no studies have assessed how non-native tree species affect cultural services in Iberia.

In this study, we propose an indicator-based approach to assess the effects of non-native trees on recreation and ecotourism, aesthetics, inspiration and cultural heritage (MEA 2005). The approach includes the use of a meta-analysis statistics, the odds ratio, to evaluate photographic, internet and catalogue data considered as relevant to infer on the effects of non-native trees in cultural ecosystem services. We apply the proposed approach at the regional level in the Iberian Peninsula (i.e. NUTS-2 administrative regions) and compare the obtained results between countries (Portugal versus Spain). Then, we evaluate if the regional variation of non-native tree effects changes along predictors related to land cover and management, socio-economy, human well-being, and climate. Finally, we provide considerations for the management of non-native trees in Iberia and discuss the potential applicability of our approach to other contexts and social-ecological challenges.

4.2. Material and methods

4.2.1. Data collection

Non-native and native tree species

We compiled information on the occurrence and abundance (represented by the cover area) of non-native and native tree species in NUTS- 2 administrative regions (Eurostat, 2015a) of the Iberian Peninsula (southwest Europe). We focused on Continental Portugal (15% of Iberian land area) and Spain, including the Balearic Islands (85% of land area). We considered the whole (introduction-)naturalisation-invasion continuum of tree species in both countries (including planted, naturalised and invasive species; Richardson and Pyšek 2006). Archeophytes and hybrids between non-native and native species were not considered. The lists of non-native trees were obtained from Almeida and Freitas (2006) for Portugal, and from Sanz Elorza et al. (2004) for Spain. The lists of native species were obtained from ICNF (2013a) for Portugal, and from Cela et al. (2013) for Spain.

In total, we considered 157 non-native and 53 native tree species for Portugal; and 261 non-native and 63 native tree species for Spain. Species nomenclature followed Castroviejo et al. (1986-2010) and was updated following The Plant List (2013). For Portugal, the area covered by non-native and native trees was obtained from the National Land Cover Map − COS 2007 (DGT, 2017) and was complemented with information from the sixth National Forest Inventory (ICNF, 2013b). For Spain, the cover area was obtained from the third National Forest Inventory − IFN3 1997-2007 (MAPAMA, 2014) and was complemented with information from Beltrán et al. (2013). Details on the lists of non-native and native tree species, and on their cover areas are shown in Appendices A and B, respectively (Supplementary material III).

Cultural ecosystem services

Grounded on the *Millennium Ecosystem Assessment* (MEA 2005), we considered four categories of cultural ecosystem services: recreation and ecotourism, aesthetics, inspiration and cultural heritage. Although other typologies for cultural services are available (e.g., *Common International Classification of Ecosystem Services* - CICES), we followed the MEA typology to allow comparability of our results with previous research on cultural services (e.g.

Hernández-Morcillo et al., 2013; Milcu et al., 2013). For each category of cultural services, we focused on distinct data types and sources. These data were selected through a participatory approach conducted under the Cost Action FP1403: *Non-native tree species for European forests − experiences, risks and opportunities* (http://nnext.boku.ac.at/). The approach involved several academics worldwide as well as literature reviews and consultations with external experts. The selection of data types and sources was made "*considering societal expression of appreciation of ecosystems (...) as a proxy for cultural ecosystem services*" (Hernández-Morcillo et al., 2013: p. 436), and relied on their cost- and time-efficiency, availability, and ease of dissemination across countries worldwide.

Our dataset was obtained through the screening of photographic, internet and catalogue information (following e.g. Hernández-Morcillo et al., 2013; Figueroa-Alfaro and Tang, 2017; Oteros-Rozas et al., 2017). For recreation and ecotourism, we focused on two data types: tourism information systems and nature routes. For tourism information systems, data sources comprised official websites of regional tourism. For nature routes, data sources included online nature routes from the "wikiloc" application (http://www.wikiloc.com). In each source, we counted the number of photographs dominated by non-native or native trees. We used a minimum threshold of 50% coverage of a tree in the photograph to be considered as dominant. Aesthetics were evaluated from two data types: catalogues of ornamental plants (online and printed catalogues of local plant dealers), and tree inventories of urban parks (available on the web, books, municipality archives, in-situ panels, and personal surveys). In each source, we counted the number of non-native and native trees. Inspiration services were assessed from collective websites on nature photography for which the location of each photograph was provided. We counted the number of photographs in which non-native or native trees were dominant. Finally, for cultural heritage we counted the number of non-native and native trees indicated in the official lists of monumental tree species of Portugal and Spain.

All data were prior to year 2016 and considered as representative of each one of the 21 NUTS-2 regions of the Iberian Peninsula. More information on data types and respective data sources is shown in Table 4.1, and further details are provided in Appendix C.

Table 4.1. Categories of cultural ecosystem services considered, with respective data types and rationale. The number of data sources (n) considered for each data type of cultural ecosystem services is shown. The table also describes the components (A-D: equations 1-4) of the indicator proposed for evaluating non-native tree effects on cultural services (see section Data analyses).

		Components of the indicator			
Data types	**Rationale**	**Frequency of non-native trees in the service (A)**	**Frequency of native trees in the service (B)**	**Frequency of non-native trees in the region (C)**	**Frequency of native trees in the region (D)**
Recreation and ecotourism					
Tourism information (n = 21)	Photographs from tourism websites have the potential to attract tourists	Number of photographs dominated by non-native trees	Number of photographs dominated by native trees	Cover of non-native trees in the region	Cover of native trees the region
Nature routes (n = 161)	Geo-referenced nature routes shared with the public translate society preferences for recreation	Number of photographs dominated by non-native trees	Number of photographs dominated by native trees	Cover of non-native trees in the region	Cover of native trees in the region
Aesthetics					
Catalogues of plant dealers (n = 28)	Tree species offered by plant dealers are appreciated mostly by ornamental values	Number of non-native tree species offered in catalogues	Number of native tree species offered in catalogues	Total number of non-native tree species in the country	Total number of native tree species in the country
Urban parks (n = 45)	Trees exhibited in urban parks are selected mostly based on their aesthetics	Number of non-native tree species present in inventories	Number of native tree species present in inventories	Total number of non-native tree species in the country	Total number of native tree species in the country
Inspiration					
Nature photographs (n = 12)	Artistic photographs reflect the choice of inspiring motifs from nature	Number of photographs dominated by non-native trees	Number of photographs dominated by native trees	Cover of non-native trees in the region	Cover of native trees in the region
Cultural heritage					
Monumental trees (n = 21)	Monumental trees are symbols of human culture, sense of place, and history	Number of non-native tree species present in the list	Number of native tree species present in the list	Cover of non-native trees in the region	Cover of native trees in the region

Environmental and socio-economic predictors

Based on previous knowledge and data availability, a first set of 24 predictors was considered to explain the observed variation of effects of non-native trees on cultural services. The predictors expressed regional patterns of land cover and management, socio-economy, human well-being and climate across the Iberian Peninsula. Land cover and management predictors derived from governmental data and cartography (ICNF, 2013b, for Portugal; MAPAMA, 2014, for Spain). Socio-economic predictors were obtained from Eurostat (2015b), with the human influence index being obtained from WCS and CIESIN (2005), and the development index from Hardeman and Dijkstra (2014). Human well-being indicators were obtained from the OECD regional well-being indices (OECD, 2013). The mean values of climatic predictors per region were calculated from the maps of the *Iberian Climate Atlas* (Ninyerola et al., 2005), using ArcGIS 10.1 (ESRI, 2012).

All continuous predictors were tested for pair-wise correlations using the non-parametric Spearman test. We excluded 12 predictors from subsequent analyses, due to correlation values above 0.60 when tested against the remaining predictors (Quinn and Keough, 2002). The final set of considered predictors is shown in Table 4.2. Details on predictors and their correlations can be found in Appendix D and E, respectively.

Table 4.2. Final set of predictors used to explain the variation of effects of non-native tree species on cultural ecosystem services across Iberian NUTS-2 regions.

Code	Predictors
Land cover and management *(Vilà and Pujadas 2001; Vicente et al. 2016)*	
Forests	Proportion of forest area
Protected areas	Proportion of protected areas
Socio-economy *(Vilà and Pujadas 2001; Krumm and Vítková 2016)*	
Country	The country where the data sources were located (Portugal or Spain)
Tourism	Number of arrivals at tourist accommodation establishments
Development	EU regional human development index (based on life expectancy, mortality, education, income, and employment)
Impact	Global human influence index (based on human settlement, accessibility, landscape transformation, and electric power infrastructures)
Human well-being *(OECD 2013; Ghosh and Traverse 2005; Vaz et al. 2017a)*	
Life	Life satisfaction, a subjective well-being index of how people evaluate their life (based on citizens' questionnaires)
Jobs	Job availability, a well-being index of material conditions (based on both employment and unemployment % rates)

Housing	Housing, an index of material conditions for well-being (based on the % ratio of the number of rooms per person)
Environment	Environmental quality, an index of human life quality (based on the estimated average exposure to air pollution in PM2.5 μg/m³)
Climate *(Gassó et al. 2009; Vicente et al. 2016)*	
Temperature	Minimum temperature of the coldest month (°C)
Precipitation	Total annual precipitation (mm)
Radiation	Annual solar radiation (W/m^2)

4.2.2. Data analyses

An indicator of non-native tree effects on cultural services

We used the term "effect" to refer to a change promoted by non-native trees on cultural ecosystem services (Jeschke et al., 2014). To describe the direction of this change, we used "increase" or "decrease" of a cultural service, respectively when non-native trees were over- or under-represented in a service (compared to native trees). By doing so, an increase or decrease of a service by non-native trees does not mean an improved or degraded state of the service (Pyšek et al., 2012).

To evaluate the effects of non-native trees, we propose an indicator based on the calculation of the odds ratio. The odds ratio is an effect size measure, often applied in meta-analysis and case-control studies, to evaluate the association between an exposure and an outcome, against the frequency of the outcome if expected by chance (Borenstein et al., 2008). In our case, the odds ratio was assumed to express the direction of effects of non-native tree species (i.e. exposure) in each data source of cultural services (i.e., outcome), compared to the effect of native trees (i.e. non-exposure or comparator). The computation of the direction of effects was further achieved considering the frequency of non-native and native trees in a data source against their frequency in the region (i.e. expected by change), as the control situation. For computing the indicator, we first organised the information of each data source (Table 4.1) in contingency tables (Table 4.3).

Table 4.3. Example of a contingency table used for calculating the indicator of non-native tree effects on cultural services, based on the odds ratio.

	Frequency of non-native trees (Exposure)	**Frequency of native trees** (Non-exposure)
Data source of cultural services (Outcome)	*A*	*B*
NUTS-2 region under analysis (Control)	*C*	*D*

For each data source, we then calculated the odds ratio in its logarithmic form (logOR), using Peto's method, since some sources showed the absence of non-native or native trees (Borenstein et al., 2008; Viechtbauer, 2010; Eqs. (1)-(4)).

Eq. (1) $\Psi = exp(O - E/V)$ (1)

Eq. (2) $O = A$ (2)

Eq. (3) $E = (A+B)/(A+C)/n$ (3)

Eq. (4) $V = (A+B)(C + D)(A + C)(B + D)/n\wedge 2(n - 1)$ (4)

In Eqs. (1)-(4), Ψ is Peto's odds ratio, and V is both weighting factor and variance for the difference between observed (O) and expected (E) values (see Appendix F for details).

Evaluating the effects of non-native trees on cultural services

For each data type of cultural services, logORs of all data sources were aggregated in a weighted logOR, using the DerSimonian-Laird random effects model. We used this model since it accounts for the variation in logOR across all sources of each data type, in addition to sampling error. Weighted logOR values higher or lower than zero respectively express an over- or under-representation of non-native trees in the cultural service, in comparison to native trees, meaning that non-native trees increase or decrease the cultural service. Weighted logOR equal to zero indicate no effect (or change) on the cultural service. We further tested whether the values obtained for each weighted logOR were significantly different from zero, through non-parametric permutation tests with 1000 iterations (Viechtbauer, 2010).

To test for significant bias in each data type, we calculated the Rosenberg fail-safe number (Rothstein et al., 2005). The fail-safe number estimates the number of additional sources that would be needed to change the results of the weighted logOR from significant to non-significant. When the fail-safe number is larger than 5N + 10 (where N is the number of data sources), the weighted logOR can be interpreted as a reliable estimate of true effects (Rothstein et al., 2005). Details on the weighted logOR computation and bias analysis are provided in Appendix G.

Testing the observed variation of non-native tree effects against predictors

For each data type of cultural services, we assessed whether the variation of non-native tree effects could be explained by the final set of predictors (see Table 4.2). The heterogeneity of logOR across all data sources (expressing the variation of non-native tree effects) was tested using the Q statistic under a chi-square distribution, with n-1 degrees of freedom (Borenstein et al., 2008; Viechtbauer, 2010). Values for the Q statistic greater than expected by sampling error suggest an underlying structure of effects in the data type (Borenstein et al., 2008).

When the Q statistic showed significant values, we performed a structured meta-analysis (Viechtbauer, 2010). Specifically, for the categorical predictor of country, we computed the weighted logOR of each data type (Peto's method under the DerSimonian-Laird random effects model) for Portugal and Spain, individually. For the continuous predictors, we used a weighted least squares regression to test for significant relations between the predictors and the values of logOR across the sources of each data type. When the regression test showed significant values, we assessed the regression slope and its significance. Significant regression values higher or lower than zero, respectively indicate that non-native trees increase or decrease the cultural service, as the predictors' values increase (Viechtbauer 2010). All statistical procedures were implemented in R software (R Core Team 2014), using the package metafor (Viechtbauer, 2010).

4.3. RESULTS

4.3.1. Effects of non-native trees on cultural ecosystem services

We found contrasting results for the weighted logOR across the different data types of cultural services in the Iberian Peninsula (Figure 4.1). Weighted logOR values higher than zero were obtained for tourism information systems (recreation and ecotourism) and for monumental trees (cultural heritage). Conversely, values lower than zero were found for nature routes (recreation and ecotourism), catalogues of plant dealers (aesthetics) and inventories of urban parks (aesthetics). Fail-safe numbers were higher than 5N + 10 (see Appendix G for full results), meaning that these significant results translate reliable estimates of non-native tree effects. No significant values were obtained for nature photographs (inspiration; Figure 4.1).

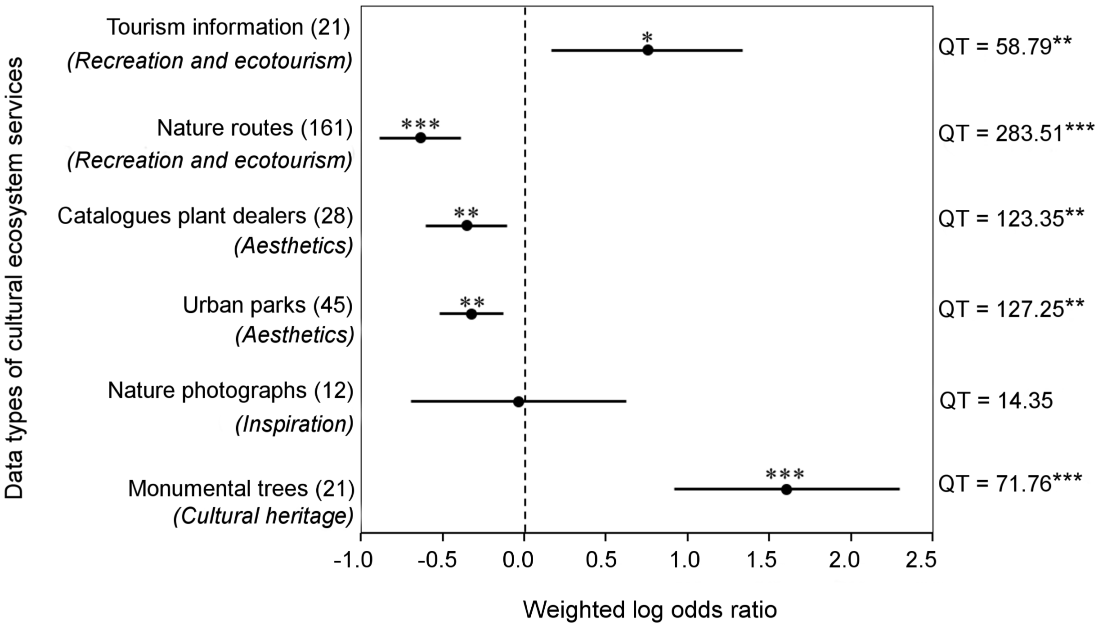

Figure 4.1. Weighted log odds ratio (Peto's method under the DerSimonian-Laird random effects model) for each data type of cultural ecosystem services (number of data sources are shown in brackets). Values higher or lower than zero respectively suggest that non-native trees increase or decrease the cultural service, in contrast to native trees. Values on the right indicate the heterogeneity (QT) of the log odds ratio across data sources of each data type, tested by means of the Q statistics. Statistical significance: *p < 0.05; **p < 0.01; ***p < 0.001.

4.3.2. Predictors of non-native tree effects on cultural ecosystem services

Significant regional variations of logOR ($p < 0.05$) were observed for most data types, except again for nature photographs (Figures 4.1 and 4.2). The categorical predictor of country (Portugal or Spain) significantly explained part of logOR variation for catalogues of plant dealers, nature routes and inventories of urban parks. Catalogues of plant dealers resulted in positive weighted logOR values for Portugal (0.47; $p < 0.05$), but negative ones for Spain (−0.48; $p < 0.01$). Negative weighted logORs were also found for nature routes and urban park inventories, but only for Spain (weighted logOR = −0.95 and −0.48; $p < 0.001$, respectively).

No significant values were found for tourism information or for monumental trees (Table 4.4; see also Appendix G for full results).

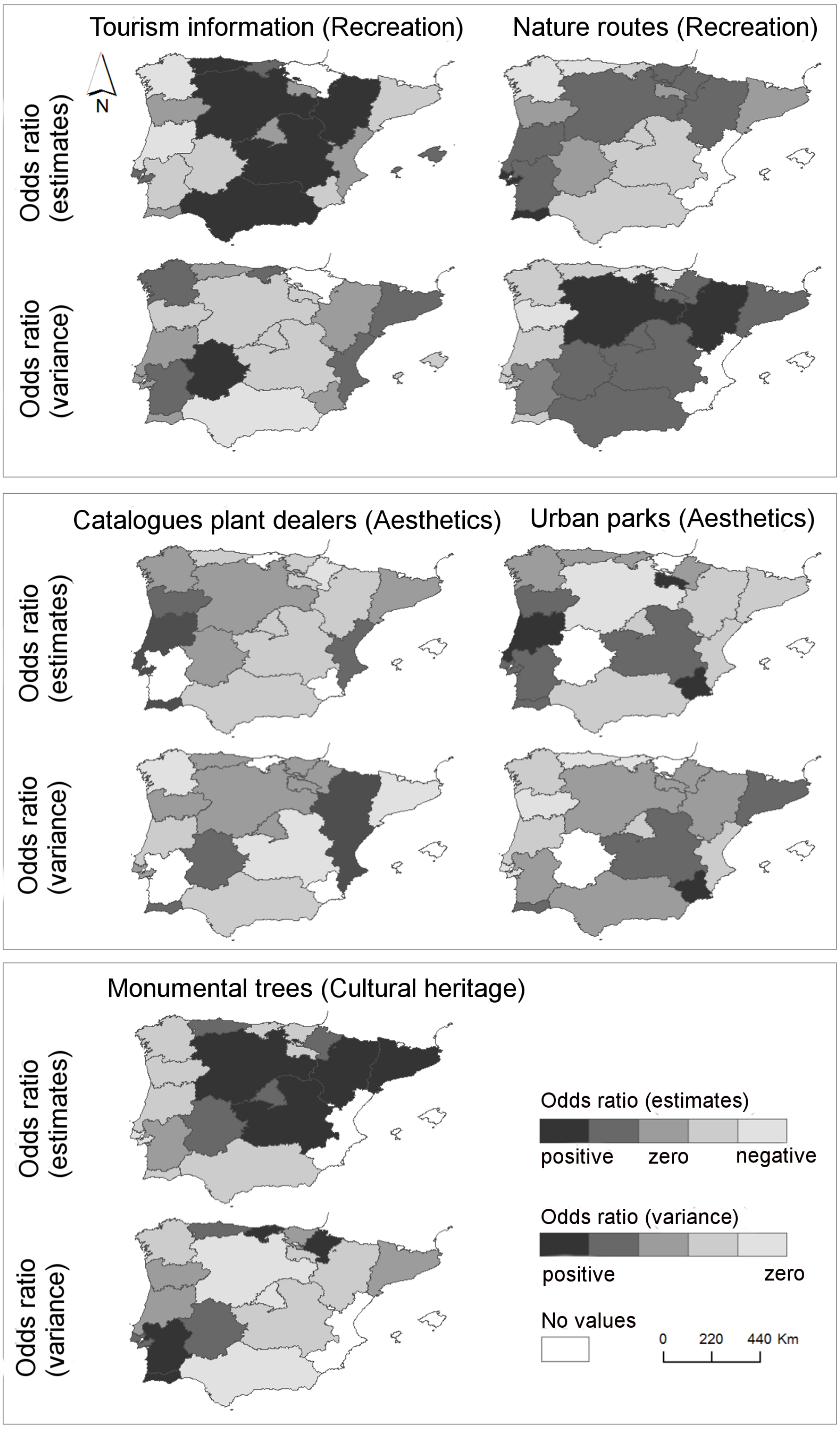

Figure 4.2. Representation of the spatial distribution of averaged estimates and variance of the log odds ratio for each Iberian NUTS-2 region. Information on nature photographs is not represented since it showed no significant weighted logOR values.

Continuous predictors (Table 4.2) also contributed to explain the variation of logOR values for most data types, except for monumental trees (Table 4.4). Job availability was negatively related to logOR values for tourism information, nature routes and urban park inventories. Life satisfaction held a negative relationship with values for nature routes, catalogues of plant dealers, and urban park inventories. Proportion of forests (negative relationship), total annual precipitation (negative) and solar radiation (positive) also explained the variation in logOR values for nature routes. Minimum temperature related positively to logOR values for catalogues of plant dealers and urban park inventories. Human development held a negative relationship with values for catalogues of plant dealers, as did tourism rates with values for urban park inventories (Table 4.4; see also Appendix H for full results).

Table 4.4. Results of the structured meta-analysis on assessing the covariation between the considered predictors (Table 4.2) and the effects of non-native trees (expressed by logORs) on data types of cultural services. The table shows the heterogeneity explained by each predictor and its significance based on a chi-square distribution with n-1 degree of freedom. Values in brackets show the regression slopes and respective significance for continuous predictors (see Appendix H for full results). Statistical significance: *p < 0.05; **p < 0.01; ***p < 0.001.

	Recreation and ecotourism		**Aesthetics**		**Cultural heritage**
	Tourism information	***Nature routes***	***Catalogues dealers***	***Urban parks***	***Monumental trees***
Land cover and management					
Forests	0.065	13.567** (-0.037**)	1.358	2.911	0.952
Protected areas	0.101	3.992	0.823	3.168	0.496
Socio-economy					
Country	9.593	52.901***	11.350*	24.152**	23.613
Tourism	2.846	2.199	0.199	11.282** (-0.001***)	0.059
Development	0.024	0.424	4.283* (-1.318*)	0.160	0.323
Impact	0.461	0.640	0.986	1.102	2.978
Human well-being					
Life	1.362	9.389** (-0.590**)	5.961** (-0.511**)	4.721* (-0.384*)	1.774
Jobs	5.063* (-0.285*)	25.257** (-0.268**)	0.366	7.497* (-0.132*)	1.289
Housing	0.024	1.616	0.025	0.001	0.256
Environment	0.000	0.223	0.489	4.737	4.083
Climate					
Temperature	0.288	1.929	4.299* (0.114*)	5.703* (0.114*)	5.301
Precipitation	0.475	5.939* (-0.0001*)	0.745	0.123	0.465
Radiation	0.031	8.178** (0.014**)	0.872	0.506	0.089

4.4. Discussion

4.4.1. Non-native tree effects on cultural ecosystem services

We developed an indicator-based approach grounded in meta-analytical techniques and applied it to evaluate the direction of effects of non-native trees on cultural ecosystem services in the Iberian Peninsula. We found that the effects of non-native trees were service-dependent, highlighting the plurality of societal preferences towards cultural ecosystem services (Chan et al., 2012; Ghosh and Traverse, 2005; Martín-López et al., 2012). We also found that the effects of non-native trees were country-dependent and determined by some environmental and socio-economic factors. Although holding common geographic and historical features, Portugal and Spain still differ in their climate, demography, politics, culture and economy. These differences could therefore influence the contribution of non-native trees to the multiple cultural services (after Humair et al., 2015; Krumm and Vítková, 2016), as previously highlighted for provisioning and regulating services (Brundu and Richardson, 2016; Carruthers et al., 2011; Kull et al., 2011).

Specifically, we found contrasting effects from non-native trees on cultural services related to recreation and ecotourism in Iberia. Non-native trees were over-represented (in comparison to native trees) in information systems ruled by official tourism entities but were under-represented in photographs from nature routes experienced by local users, particularly in Spain. In the case of official entities, publicity on Iberian touristic destinations tends to show photographs covering iconic standard features from nature (Santos, 2004), which may include non-native species (e.g. palm trees in coastal areas, pines or sequoias in forest areas). Nature route users, however, may enjoy landscapes with more pristine nature features. In both Portugal and Spain, many areas are dominated by non-native trees (e.g., *Eucalyptus globulus*, *Pinus radiata*, *Robinia pseudoacacia*, or *Acacia* spp.), producing monotonous and homogeneous landscapes that can be less attractive to people (following Humair et al., 2015; Kueffer and Kull, 2017; Richardson et al., 2014).

We found that non-native trees decreased aesthetic services in the Iberian Peninsula, and particularly in Spain. Still, we found no significant effects of non-natives on the pool of tree species in urban parks in Portugal. This is in contrast to Spain, where legal considerations on the adoption of non-native trees have been explicitly taken for urban areas (Royal Decree-Law 630/2013: 5th disposition). We also found an increase in aesthetic services by non-native trees when focusing on catalogues of plant dealers in Portugal (in contrast to Spain), suggesting a market preference for these species. This is of relevance considering that these catalogues include sets of ornamental plants commonly traded in horticulture. Despite legal constraints on the trade of non-native species in both countries (Decree-Laws 565/99 and 630/2013; EU Regulation 1143/2014), horticultural trade is still a main introduction pathway and distribution channel of non-native plants that may become invasive (Hulme et al., 2017; Humair et al., 2015). This is often due to a lack of awareness and information on the non-nativeness of traded ornamental species among sellers, customers, and regulatory entities (Andreu et al., 2009; Carruthers et al., 2011).

We found no significant effects of non-native tree species on inspiration cultural services. This result suggests that the notion of species nativeness in Iberia (i.e. non-natives versus natives) may not influence inspirational preferences of the public and hence photographers, as previously highlighted by Oteros-Rozas et al. (2017) and by van Berkel and Verburg (2014) for rural landscapes. Nevertheless, the non-significance of our result can also express the limited number of observations in our test area. Whenever available, complementary data sources should be explored, namely art museum databases and catalogues, photography literature, and other social media (e.g., Flickr, Panoramio; Figueroa-Alfaro and Tang, 2017).

Non-native trees increased cultural heritage services in the Iberian Peninsula, expressed by the over-representation of non-native trees (compared to native trees) in official lists of monumental trees. Monumental trees are part of the cultural heritage at regional and national levels, often representing symbols of human identity for local communities (Asciuto et al., 2015; Crews, 2003). In both Portugal and Spain, the monumental status of a tree can be declared due to historical backgrounds, regardless of a native or non-native status (Decree-Law 53/2012: Ordinance 124/2014). The over-representation of non-native trees in this service may express the fact that many non-native trees became monumental trees in Iberia after being introduced as botanical curiosities or research assets during past transatlantic expeditions (e.g., *Camellia japonica* L.), or due to their long-term economic symbolism (e.g., *Eucalyptus globulus*; see also Asciuto et al., 2015; Crews, 2003).

4.4.2. Predictors of non-native tree effects: considerations for management

We found higher increases in recreation and ecotourism, and aesthetic services by non-native trees in NUTS-2 regions with lower socio-economic conditions (tourism rates, development level, job availability) and lower life satisfaction levels. Developed countries are known to host more non-native plant species than developing ones (Humair et al., 2015; Vilà and Pujadas, 2001). Our results add that non-native trees seem to be more used (than native trees) for aesthetic and recreational purposes in less developed regions (i.e. under lower income and educational levels). A higher use of non-natives in these regions may be due, not only to intrinsic preferences by people, but also to lower awareness on the notion of non-native trees and related risks (following Carruthers et al., 2011; Hulme et al., 2017; Kueffer and Kull, 2017). Non-native trees also contributed more to the former services in warmer and drier regions with less forested land. In Iberia, these less developed regions are mostly under warmer and drier climates and hold fewer forested areas. This may be of importance considering that climate change is expected to increase the likelihood of naturalisation for some ornamental plants, and thus their capacity to alter cultural (and other) ecosystem services (Dullinger et al., 2017; see also Seebens et al., 2015).

The effects of non-native trees on inspiration services and cultural heritage were, however, not explained by the considered predictors. As highlighted by van Berkel and Verburg (2014) and Kueffer and Kull (2017), inspirational and heritage values of non-native trees can also relate to e.g. long-term associations between people and species, human traditions, affections and interests, and symbolic representations of nature, which are difficult to assess outside their regional context. Therefore, our results suggest that the considered social-ecological context may not be of significant relevance for inspirational and heritage services of non-native trees in Iberia, highlighting the need to further explore human psychological and cognitive factors, which were not available for our analysis.

Overall, our results highlight four main ideas to be considered in the management of non-native trees in Iberia. First, the effects of non-native trees on cultural services depend on people's preferences towards visual features. In Iberia, visual attributes of non-native trees can be widely associated to homogenised and monotonous landscapes (Kueffer and Kull 2017), explaining the lower consideration of these species for recreation and ecotourism by the general public, but not by official tourism entities. Second, the idea of "out-of-normal" features, as well as of testimonies of historical and cultural events, can be attributed to non-native tree species (Carruthers et al., 2011; Crews 2003). In Portugal and Spain, this can justify the consideration of non-native trees as attractions for recreation and ecotourism by official tourism entities, and as monumental assets in cultural heritage. Third, awareness of the notion of “non-native” associated to tree species depends on the social-ecological context (Kueffer and Kull, 2017), and it can influence the ornamental and market value of potentially traded species. Fourth, people from developed socio-economic (including educational) contexts may be more aware of risks associated to non-native species (Vilà and Pujadas, 2001; Marchante and Marchante, 2016). In Iberia, this can explain why we found a higher contribution of non-native trees to cultural services in less developed regions.

We suggest that management strategies targeting non-native trees should promote awareness, e.g. by means of environmental education programmes, public outreach and further information campaigns (Marchante and Marchante, 2016). In Iberia, these campaigns should prioritise tourism entities and ornamental trade, especially in less developed regions. Biosecurity efforts should thus be reinforced among managers, sellers and local residents, who influence interactions among non-native species, social media and market values (Hulme et al., 2017; Humair et al., 2015; Marchante and Marchante, 2016). Also, since our research considered non-native trees as a whole, local human perceptions towards individual species should be further considered, as they may differ among species and regions (Kueffer and Kull, 2017). Researchers and managers should further examine the motivations underlying the choices and preferences towards non-native ornamental trees (Hulme et al., 2017; Seebens et al., 2015). Promoting risk awareness and strengthening biosecurity efforts, specially focusing on the fact that some of non-natives may naturalise and become invasive (e.g., *Acacia longifolia, Pseudotsuga menziesii* and *Robinia pseudoacacia*), could prevent undesirable alterations on ecosystem services (Andreu et al., 2009; Hulme et al., 2017 Vaz et al., 2017a,b).

4.4.3. Methodological considerations

We proposed an indicator-based approach to obtain preliminary insights on the direction of effects of non-native tree species on cultural services in the Iberia Peninsula. The proposed approach is able to integrate multiple data types from widely available sources of cultural services, allowing reproducibility and the inclusion of further information as data sources expand (Zhang et al., 2016). The approach also has the potential to be applicable to other taxonomic groups, biodiversity measures (e.g. abundance), social-ecological drivers (e.g. pre- and post-invasion processes) and challenges (e.g. ecosystem disservices), and further temporal and spatial scales (Blicharska et al., 2017; Hernández-Morcillo et al., 2013; Schröter et al., 2016).

Nevertheless, the odds ratio methodology also has some constrains, as it might be sensitive to the choice of data types and control data (represented in our study by the proportion of native and non-native trees in each NUTS-2 region). Despite our study considered the most relevant and available data to quantify the relations between non- native trees and cultural ecosystem services, we still encourage the study of complementary types and sources of information (following e.g., Figueroa-Alfaro and Tang, 2017; Oteros-Rozas et al., 2017; van Berkel and Verburg, 2014). Particular attention could be given to data types related to inspirational services that did not show significant results in our study. Future studies should also examine information at different time periods and geographic areas, targeting other social-ecological challenges, and consider practical ways to validate results in specific contexts (Hernández-Morcillo et al., 2013; Milcu et al., 2013).

4.5. Conclusions

We proposed an indicator-based approach to analyse patterns and drivers of cultural ecosystem services. The methodology combines meta-analytical techniques with the collection of different types of information from multiple sources. We applied this approach to the Iberian Peninsula to evaluate the effects of non-native trees on cultural services i.e., on recreation and ecotourism, aesthetics, inspiration and cultural heritage. Those effects differed among services and countries.

In short, non-native trees increased recreation and ecotourism services, when focusing on photographs from official tourism entities, but not from nature route users. Data from inventories of urban parks and catalogues of ornamental plant dealers suggest that non-native trees decreased aesthetics services, particularly in Spain and in contrast to Portugal. We also found an increase of cultural heritage services, expressed by an over-representation of non-native trees (compared to native trees) in catalogues of monumental trees. However, no significant effects were observed on inspiration services. Overall, higher increases of cultural services by non-native trees were observed in less developed regions (i.e., under lower income and educational levels) with lower life satisfaction indices.

Our approach and our results provide pioneer insights into the cultural dimension of non-native trees in Iberia. We recommend that management and biosecurity actions should promote awareness and outreach campaigns on non-native trees. A special focus should be provided to official entities of regional tourism and to ornamental plant dealers, as well as customers and authorities, especially in less developed regions. Finally, we call for studies that expand the proposed approach and explore the role of further global change processes on (cultural) ecosystem services.

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SUPPLEMENTARY MATERIAL III

Appendix A - Information on the lists of non-native and native tree species

We followed Richardson and Rejmánek (2011, p: 789) to “define trees as perennial woody plants with many secondary branches supported clear of the ground on a single main stem or trunk with clear apical dominance (we added palms which are usually considered trees).” We included small trees, as perennial woody plants developing an arboreal structure with a distinguishable bole, even if multi-stemmed. Non-native tree species were defined as those tree species that were introduced (accidentally or intentionally) by humans to new geographic areas (following Richardson et al., 2011). Archeophytes were excluded from our dataset, i.e., non-native tree species for which there is evidence showing their establishment in the Iberian Peninsula before the year of 1492, when the Columbian exchange began. We are aware that the lists may include cryptogenic species (e.g., *Cupressus sempervirens* L., or some Prunus species) suspected of being archaeophytes. However, due to the absence of clear evidence or consensus in scientific literature, these species were maintained according to their official native or non-native status for Portugal (ICNF, 2013a,b) and Spain (Sanz Elorza et al., 2004).

The native or non-native status of a tree species was stated at species level (and not at the subspecies or variety level) and at the country scale. This allowed avoiding the problem of having a tree considered as non-native tree species in a country’s continent but as native tree species in the same country’s islands (and vice versa). Also, for this reason, oceanic islands (i.e., Azores, Madera and Canary Islands) were not considered in this study.

For Spain, the list of non-native tree species was derived from Sanz Elorza et al. (2004), and was complemented with information from urban parks, catalogues of ornamental plant dealers, and monumental trees. The list of native tree species was obtained from “Arboles Ibéricos” (http://www.arbolesibericos.es/). For Portugal, the list of non-native tree species was based on Almeida and Freitas (2006) and was complemented with information from catalogues of plant dealers, urban parks, Pereira et al. (2016), and official information on exotic species for Portugal (http://www.icnf.pt/portal/naturaclas/patrinatur/especies/n-indig/n-ind). The list of native tree species was extracted from ICNF (ICNF, 2013a,b). Both non-native and native tree species lists are shown in Table S4.1. Unresolved scientific names were excluded. Hybrids were considered when both parental species were native tree or non-native tree species; thus, native x non-native tree species hybrids were excluded. The lists were checked by two plant taxonomists Carlos Vila-Viçosa (CIBIO-InBIO/University of Porto), and Paulo Alves (FloraData). Species nomenclature follows Flora Iberica (Castroviejo et al., 2010) and is updated based on The Plant List (2013).

Table S4.1. List of non-native tree species (NNT) and native tree species (NT) considered for Portugal and Spain.

Species name	**Portugal**	**Spain**
Abies alba Mill.	NNT	NT
Abies concolor (Gordon) Lindl. ex Hildebr.		NNT
Abies koreana E.H.Wilson	NNT	NNT
Abies nordmanniana (Steven) Spach	NNT	NNT
Abies pinsapo Boiss.	NNT	NT
Abies procera Rehder		NNT
Abies x majoannis (*Abies alba* Mill. *x Abies pinsapo* Boiss.)		NT
Acacia baileyana F.Muell.	NNT	NNT
Acacia cultriformis G.Don	NNT	
Acacia cyclops G.Don	NNT	
Acacia dealbata Link	NNT	NNT
Acacia decurrens (J. C. Wendl.) Willd.	NNT	
Acacia floribunda (Vent.) Willd.		NNT
Acacia karroo Hayne	NNT	
Acacia longifolia (Andrews) Willd.	NNT	NNT
Acacia mearnsii De Wild.	NNT	
Acacia melanoxylon R. Br	NNT	
Acacia pendula G.Don		NNT
Acacia pycnantha Bentham	NNT	
Acacia saligna (Labill.) H.L. Wendl.	NNT	NNT
Acacia verticillata (L' Hér.) Willd.	NNT	
Acca sellowiana (O.Berg) Burret	NNT	
Acer buergerianum Miq.		NNT
Acer campestre L.	NNT	NT
Acer cappadocicum Gled.	NNT	NNT
Acer davidii Franch.		NNT
Acer japonicum Thunb.	NNT	NNT
Acer monspessulanum L.	NT	NT
Acer negundo L.	NNT	NNT
Acer opalus Mill.		NT
Acer palmatum Thunb.	NNT	NNT
Acer platanoides L.	NNT	NT
Acer pseudoplatanus L.	NT	NT
Acer rubrum L.	NNT	NNT
Acer saccharum Marshall	NNT	NNT
Acer tataricum L.	NNT	

Acer x freemanii A.E.Murray		NNT
Aesculus hippocastanum L.	NNT	NNT
Ailanthus altissima (Mill.) Swingle	NNT	NNT
Albizia julibrissin Durazz.	NNT	NNT
Alnus cordata (Loisel.) Duby	NNT	NNT
Alnus glutinosa (L.) Gaertn.	NT	NT
Alnus incana (L.) Moench	NNT	
Amelanchier ovalis Medik.	NT	
Anacardium occidentale L.		NNT
Annona cherimola Mill.		NNT
Araucaria araucana (Molina) K.Koch		NNT
Araucaria columnaris (G.Forst.) Hook.	NNT	
Araucaria heterophylla (Salisb.) Franco	NNT	NNT
Arbutus unedo L.	NT	NT
Archontophoenix alexandrae (F.Muell.) H.Wendl. & Drude		NNT
Archontophoenix cunninghamiana (H.Wendl.) H.Wendl. & Drude		NNT
Archontophoenix purpurea Hodel & Dowe		NNT
Balantium antarcticum (Labill.) C. Presl		NNT
Banksia integrifolia L.f.		NNT
Beaucarnea recurvata Lem.	NNT	NNT
Betula nigra L.		NNT
Betula papyrifera Marshall.	NNT	NNT
Betula pendula Roth.	NNT	NT
Betula pubescens Ehrh.	NT	NT
Betula utilis D.Don	NNT	NNT
Bismarckia nobilis Hildebr. & H.Wendl.		NNT
Boswellia sacra Flueck.		NNT
Brachychiton acerifolius (A.Cunn. ex G.Don) F.Muell.	NNT	NNT
Brachychiton bidwillii Hook.		NNT
Brachychiton discolor F.Muell.		NNT
Brachychiton populneus (Schott & Endl.) R.Br.	NNT	NNT
Brachychiton rupestris (T.Mitch. ex Lindl.) K.Schum.		NNT
Brahea armata S.Watson		NNT
Brahea edulis H.Wendl. ex S.Watson		NNT
Broussonetia papyrifera (L.) Vent	NNT	NNT
Butia capitata (Mart.) Becc.		NNT
Butia eriospatha (Mart. ex Drude) Becc.		NNT
Butia yatay (Mart.) Becc.		NNT
Callistemon viminalis (Sol. ex Gaertn.) G.Don		NNT
Calocedrus decurrens (Torr.) Florin		NNT
Camellia japonica L.	NNT	NNT
Carpinus betulus L.	NNT	NNT

Carya illinoinensis (Wangenh.) K.Koch		NNT
Caryota maxima Blume ex Mart.		NNT
Caryota mitis Lour.		NNT
Caryota urens L.		NNT
Casimiroa edulis La Llave		NNT
Castanea x neglecta Dode	NNT	
Casuarina equisetifolia L.	NNT	NNT
Catalpa bignonioides Walter	NNT	NNT
Catalpa bungei C.A.Mey.	NNT	NNT
Cedrus atlantica (Endl.) Manetti ex Carrière	NNT	NNT
Cedrus deodara (Roxb. ex D.Don) G.Don	NNT	NNT
Cedrus libani A.Rich.	NNT	NNT
Ceiba speciosa (A.St.-Hil.) Ravenna		NNT
Celtis australis L.	NT	NT
Celtis occidentalis L.	NNT	NNT
Cephalotaxus harringtonii (Knight ex J.Forbes) K.Koch		NNT
Cercidiphyllum japonicum Siebold & Zucc. ex J.J.Hoffm. & J.H.Schult.bis		NNT
Cercis canadensis L.		NNT
Cercis siliquastrum L.	NNT	NNT
Chamaecyparis lawsoniana (A.Murray bis) Parl.	NNT	NNT
Chamaecyparis obtusa (Siebold & Zucc.) Endl.	NNT	NNT
Chamaecyparis pisifera (Siebold & Zucc.) Endl.	NNT	NNT
Chamaecyparis thyoides (L.) Britton, Sterns & Poggenb.		NNT
Chambeyronia macrocarpa (Brongn.) Vieill. ex Becc.		NNT
Cinnamomum camphora (L.) J.Presl	NNT	NNT
Citronella mucronata (Ruiz & Pav.) D.Don		NNT
Citrus reticulata Blanco		NNT
Citrus sinensis (L.) Osbeck		NNT
Cleyera japonica Thunb.		NNT
Copernicia alba Morong		NNT
Cordyline australis (G.Forst.) Endl.		NNT
Cordyline indivisa (G.Forst.) Endl.		NNT
Cornus capitata Wall.	NNT	
Cornus controversa Hemsl.		NNT
Cornus florida L.	NNT	NNT
Cornus kousa F.Buerger ex Hance		NNT
Corylus colurna L.		NNT
Corymbia citriodora (Hook.) K.D.Hill & L.A.S.Johnson		NNT
Corymbia ficifolia (F.Muell.) K.D.Hill & L.A.S.Johnson		NNT
Crataegus monogyna Jacq.	NT	NT
Cryptomeria japonica (Thunb. ex L.f.) D.Don	NNT	NNT
Cupressus arizonica Greene	NNT	NNT

Cupressus lusitanica Mill.	NNT	NNT
Cupressus macrocarpa Hartw.	NNT	NNT
Cupressus nootkatensis D.Don		NNT
Cupressus sempervirens L.	NNT	NNT
Cussonia spicata Thunb.		NNT
Cycas circinalis L.		NNT
Cycas rumphii Miq.		NNT
Davidia involucrata Baill.	NNT	NNT
Delonix regia (Hook.) Raf.		NNT
Diospyros kaki L.f.		NNT
Dombeya tiliacea (Endl.) Planch.		NNT
Dracaena draco (L.) L.	NNT	NNT
Dypsis decaryi (Jum.) Beentje & J.Dransf.		NNT
Dypsis decipiens (Becc.) Beentje & J.Dransf.		NNT
Elaeagnus angustifolia L.	NNT	NNT
Eriobotrya japonica (Thunb.) Lindl.		NNT
Erythrina caffra Thunb.	NNT	NNT
Erythrina crista-galli L.	NNT	NNT
Erythrina falcata Benth.	NNT	NNT
Eucalyptus camaldulensis Dehnh.	NNT	NNT
Eucalyptus coccifera Hook.f.		NNT
Eucalyptus diversicolor F.Muell.	NNT	
Eucalyptus ficifolia F.Muell.	NNT	
Eucalyptus globulus Labill.	NNT	NNT
Eucalyptus gunni Hook.f.	NNT	NNT
Eucalyptus nitens (H.Deane & Maiden) Maiden	NNT	NNT
Eucalyptus parvula L.A.S.Johnson & K.D.Hill		NNT
Eucalyptus pulverulenta Sims		NNT
Eucalyptus robusta Sm.	NNT	
Eucalyptus sideroxylon A. Cunn	NNT	NNT
Eugenia brasiliensis Lam.		NNT
Eugenia uniflora L.		NNT
Euonymus japonicus Thunb.	NNT	NNT
Fagus sylvatica L.	NNT	NT
Ficus benjamina L.	NNT	NNT
Ficus elastica Roxb. ex Hornem.	NNT	NNT
Ficus macrophylla Desf. ex Pers.		NNT
Ficus microcarpa L.f.		NNT
Ficus rubiginosa Desf. ex Vent.		NNT
Firmiana simplex (L.) W.Wight		NNT
Frangula alnus Mill.	NT	NT
Fraxinus americana L.	NNT	NNT

Fraxinus angustifolia Vahl	NT	NT
Fraxinus excelsior L.	NNT	NT
Fraxinus ornus L.	NNT	
Ginkgo biloba L.	NNT	NNT
Gleditsia triacanthos L.	NNT	NNT
Grevillea robusta A.Cunn. ex R.Br.	NNT	NNT
Handroanthus chrysanthus (Jacq.) S.O.Grose		NNT
Ilex aquifolium L.	NT	NT
Jacaranda mimosifolia D.Don	NNT	NNT
Jubaea chilensis (Molina) Baill.		NNT
Juglans nigra L.	NNT	NNT
Juniperus chinensis L.		NNT
Juniperus communis L.	NT	NT
Juniperus oxycedrus L.	NT	NT
Juniperus phoenicea L.	NT	NT
Juniperus scopulorum Sarg.	NNT	NNT
Juniperus thurifera L.		NT
Juniperus virginiana L.	NNT	NNT
Koelreuteria bipinnata Franch.		NNT
Koelreuteria paniculata Laxm.	NNT	NNT
Laburnum anagyroides Medik.	NNT	NNT
Lagerstroemia indica L.	NNT	NNT
Lagunaria patersonii (Andrews) G. Don	NNT	NNT
Larix decidua Mill.		NNT
Laurus nobilis L.	NT	NT
Leucaena leucocephala (Lam.) de Wit	NNT	NNT
Ligustrum japonicum Thunb.	NNT	NNT
Ligustrum lucidum Aiton	NNT	NNT
Liquidambar acalycina H.T.Chang		NNT
Liquidambar styraciflua L.	NNT	NNT
Liriodendron tulipifera L.	NNT	NNT
Litchi chinensis Sonn.		NNT
Livistona australis (R.Br.) Mart.		NNT
Livistona chinensis (Jacq.) R.Br. ex Mart.		NNT
Macadamia integrifolia Maiden & Betche		NNT
Magnolia denudata Desr.	NNT	NNT
Magnolia grandiflora L.	NNT	NNT
Magnolia kobus DC.	NNT	NNT
Magnolia stellata (Siebold & Zucc.) Maxim.		NNT
Magnolia x soulangeana (L.)L.	NNT	NNT
Malus floribunda Siebold ex Van Houtte		NNT
Malus sylvestris (L.) Mill.	NT	NT

Mangifera indica L.		NNT
Melaleuca ericifolia Sm.	NNT	NNT
Melia azedarach L.	NNT	NNT
Mespilus germanica L.		NNT
Metasequoia glyptostroboides Hu & W.C.Cheng		NNT
Metrosideros excelsa Sol. ex Gaertn.	NNT	NNT
Morus alba L.	NNT	NNT
Morus australis Poir.		NNT
Morus nigra L.	NNT	
Myrica faya Dryand.	NT	
Nerium oleander L.	NT	NT
Nyssa sylvatica Marshall		NNT
Ostrya carpinifolia Scop.		NNT
Pachypodium lamerei Drake		NNT
Pandanus utilis Bory		NNT
Parajubaea cocoides Burret		NNT
Parajubaea torallyi (Mart.) Burret		NNT
Paulownia tomentosa Steud.	NNT	NNT
Phanera purpurea (L.) Benth.		NNT
Phanera variegata (L.) Benth.		NNT
Phoenix canariensis Chabaud	NNT	NNT
Phoenix reclinata Jacq.		NNT
Phoenix roebelenii O'Brien		NNT
Phoenix theophrasti Greuter		NNT
Phytolacca dioica L.	NNT	NNT
Picea abies (L.) H.Karst.	NNT	NNT
*Picea glauca (*Moench) Voss	NNT	NNT
Picea koraiensis Nakai		NNT
Picea omorika (Pancic) Purk.		NNT
Picea pungens Engelm.		NNT
Picea sitchensis (Bong.) Carrière	NNT	
Pinus brutia Ten.		NNT
Pinus canariensis C.Sm.		NT
Pinus densiflora Siebold & Zucc.		NNT
Pinus halepensis Mill.	NNT	NT
Pinus heldreichii Christ		NNT
Pinus mugo Turra	NNT	NNT
Pinus nigra J.F.Arnold s.l.	NNT	NNT
Pinus palustris Mill.		NNT
Pinus patula Schiede ex Schltdl. & Cham.		NNT
Pinus pinaster Ainton	NT	NT
Pinus radiata D.Don	NNT	NNT

Pinus strobus L.	NNT	NNT
Pinus sylvestris L.	NT	NT
Pinus wallichiana A.B.Jacks.	NNT	
Pistacia atlantica Desf.	NNT	NNT
Pistacia terebinthus L.	NT	NT
Pistacia vera L.		NNT
Pittosporum undulatum Vent	NNT	
Platanus acerifolia (Aiton) Willd.	NNT	
Platanus hispanica Miller ex Münchh.	NNT	NNT
Platanus orientalis L.	NNT	NNT
Platycladus orientalis (L.) Franco		NNT
Plumeria rubra L.		NNT
Populus alba L.	NT	NT
Populus deltoides Marshall	NNT	
Populus simonii Carrière		NNT
Populus tremula L.	NT	NT
Pritchardia hillebrandii Becc.		NNT
Prunus armeniaca L.		NNT
Prunus avium (L.) L.	NT	NT
Prunus cerasifera Ehrh.	NNT	NNT
Prunus incisa Thunb.		NNT
Prunus laurocerasus L.		NNT
Prunus lusitanica L.	NT	NT
Prunus mahaleb L.	NT	
Prunus padus L.	NT	NT
Prunus serotina Ehrh.	NNT	NNT
Prunus serrulata Lindl.	NNT	NNT
Prunus spinosa L.	NT	
Prunus subhirtella Miq.	NNT	NNT
Prunus yedoensis Matsum.		NNT
Pseudophoenix sargentii H.Wendl. ex Sarg.		NNT
Pseudotsuga menziesii (Mirb.) Franco	NNT	NNT
Psidium cattleianum Afzel. ex Sabine		NNT
Pyrus bourgaeana Decne.	NT	NT
Pyrus calleryana Decne.	NNT	NNT
Pyrus communis L.		NNT
Pyrus cordata Desv.	NT	NT
Pyrus pyraster (L.) Burgsd.	NT	NT
Pyrus pyrifolia (Burm.f.) Nakai		NNT
Pyrus salicifolia Pall.		NNT
Quercus canariensis Willd.	NT	
Quercus cerris L.		NNT

Quercus coccinea Münchh.	NNT	
Quercus faginea Lam.	NT	NT
Quercus palustris Münchh.	NNT	NNT
Quercus petraea (Matt.) Liebl.	NT	NT
Quercus pubescens Willd.	NT	NT
Quercus pyrenaica Willd.	NT	NT
Quercus rivasmartinezii (Capelo & J.C.Costa) Capelo & J.C.Costa	NT	NT
Quercus robur L.	NT	NT
Quercus rotundifolia Lam.	NT	NT
Quercus rubra L.	NNT	NNT
Quercus suber L.	NT	NT
Radermachera sinica (Hance) Hemsl.		NNT
Ravenea rivularis Jum. & H.Perrier		NNT
Rhus typhina L.		NNT
Robinia margarettae Ashe	NNT	
Robinia pseudoacacia L.	NNT	NNT
Roystonea regia (Kunth) O.F.Cook		NNT
Sabal mexicana Mart.		NNT
Sabal palmetto (Walter) Lodd. ex Schult. & Schult.f.		NNT
Salix alba L.	NT	NT
Salix atrocinerea Brot.	NT	NT
Salix babylonica L.	NNT	NNT
Salix caprea L.	NT	NT
Salix fragilis L.	NNT	NT
Salix humboldtiana Willd.		NNT
Salix pentandra L.		NNT
Salix salviifolia Brot.	NT	
Salix triandra L.	NT	NT
Sambucus nigra L.	NT	NT
Schinus molle L.	NNT	NNT
Schinus terebinthifolia Raddi	NNT	NNT
Sequoia sempervirens (D.Don) Endl.		NNT
Sequoiadendron giganteum (Lindl.) J.Buchholz	NNT	NNT
Sorbus aria (L.) Crantz	NT	NT
Sorbus aucuparia L.	NT	NT
Sorbus intermedia (Ehrh.) Pers.		NNT
Sorbus latifolia (Lam.) Pers.	NT	NT
Sorbus torminalis (L.) Crantz	NT	NT
Spathodea campanulata P.Beauv.		NNT
Stenocarpus sinuatus (A. Cunn.) Endl.		NNT
Styphnolobium japonicum (L.) Schott	NNT	NNT
Syagrus romanzoffiana (Cham.) Glassman		NNT

Syringa vulgaris L.	NNT	NNT
Tamarix africana Poir.	NT	NT
Tamarix canariensis Willd.	NT	
Taxodium distichum (L.) Rich.	NNT	NNT
Taxus baccata L.	NT	NT
Tetraclinis articulata (Vahl) Mast.		NT
Thuja occidentalis L.	NNT	NNT
Thuja plicata Donn ex D.Don	NNT	NNT
Tilia americana L.	NNT	NNT
Tilia cordata Mill.	NNT	NT
Tilia henryana Szyszył.	NNT	
Tilia mongolica Maxim.	NNT	
Tilia platyphyllos Scop.	NNT	NT
Tilia tomentosa Moench	NNT	NNT
Tipuana tipu (Benth.) Kuntze	NNT	NNT
Trachycarpus fortunei (Hook.) H.Wendl.		NNT
Trachycarpus martianus (Wall. ex Mart.) H.Wendl.		NNT
Ulmus glabra Huds.	NNT	NT
Ulmus laevis Pall.		NNT
Ulmus pumila L.	NNT	NNT
Veitchia joannis H.Wendl.		NNT
Washingtonia filifera (Linden ex André) H.Wendl. ex de Bary	NNT	NNT
Washingtonia robusta H.Wendl.		NNT
Wodyetia bifurcata A.K.Irvine		NNT
Yucca gigantea Lem.		NNT
Yucca rostrata Engelm. ex Trel.		NNT
Zelkova carpinifolia (Pall.) K. Koch		NNT
Zelkova serrata (Thunb.) Makino		NNT
Ziziphus jujuba Mill.		NNT

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Appendix B - The calculation of cover areas of non-native and native tree species

This information concerns the calculation of the areas covered by non-native and native tree species in each region of Portugal and Spain. The areas were considered in the following data types: “wikiloc nature routes”, “official tourism websites”, “monumental tree species”, and “artistic nature photographs”.

Spanish data were obtained from the 3rd National Forest Inventory (IFN3 1997-2007), available at the Ministry of Agriculture, Fishing, Food and the Environment webpage (http://www.mapama.gob.es/es/biodiversidad/servicios/banco-datos-naturaleza/informacion-disponible/ifn3.aspx). Data are provided at the province scale (i.e., NUTS-2 regions), and the units express the basimetric area (sum of all section areas of trees at 1.30 m height, referred to a hectare). The IFN only covers large forest areas, so species present in small patches or narrow corridors (e.g., riparian forests) may not be included. Because of this, 7 provinces had an area of non-native tree species area of zero. To get information on these provinces, we used an additional source (Beltrán et al. 2013), that provides the cover area (in hectares) of native and non-native forest types. This additional source allowed fulfilling 4 out of the 7 regions with zero-non-native tree species-cover. We used the non-native /native tree species cover proportion of these 4 regions to calculate the non-native tree cover in IFN units. The remaining regions with zero values (i.e. Murcia, Baleares, and Valencia) were kept with zero, and were not considered in subsequent analyses.

Portuguese data were obtained from COS2007 (available at: http://www.dgterritorio.pt/cartografia_e_geodesia/cartografia/cartografia_tematica/carta_de_ocupacao_do_solo__cos_/cos__2007/). This source includes spatial data provided as shapefile for the country, in hectares (ha). Since the land cover data is not always provided at the species level, we considered a conservative approach in the sense that only classes dominated by non-native or native tree species land cover were considered. To do so, we followed the categories indicated in Table S4.2. To assess the area covered by native and non-native tree species per NUTS-2 region, the information from COS2007 was merged with the shapefile for the administrative regions from Eurostat (available at: http://ec.europa.eu/eurostat/web/gisco/geodata/reference-data/administrative-units-statistical-units). The final values were compared to the values provided by the National Forest Inventory (ICNF 2013), and the obtained proportions of non-native/native tree species cover areas were validated.

Table S4.2. Description of the land cover levels available in COS 2007 with indication of the native (NT) and non-native (NNT) type of land-cover.

Level 3	**Level 4**	**Level 5**	**Description**	**NT/NNT**
Agro-forestry systems (2.4.4)				
2.4.4	2.4.4.01	2.4.4.01.1	*Quercus suber* w/ dry cultures	NT
2.4.4	2.4.4.01	2.4.4.01.2	*Q. ilex* w/ dry cultures	NT
2.4.4	2.4.4.01	2.4.4.01.2	Other *Quercus* w/ dry cultures	NT
2.4.4	2.4.4.01	2.4.4.01.5	*Q. suber* and *Q. ilex* w/ dry cultures	NT
2.4.4	2.4.4.02	2.4.4.02.1	*Q. suber* w/ irrigated cultures	NT
2.4.4	2.4.4.02	2.4.4.02.2	*Q. ilex* w/ irrigated cultures	NT
2.4.4	2.4.4.02	2.4.4.02.3	Other *Quercus* w/ irrigated cultures	NT
2.4.4	2.4.4.02	2.4.4.02.5	*Q. suber* and *Q. ilex* w/ irrigated cultures	NT
2.4.4	2.4.4.03	2.4.4.03.1	*Q. suber* w/ pastures	NT
2.4.4	2.4.4.03	2.4.4.03.2	*Q. ilex* w/ pastures	NT
2.4.4	2.4.4.03	2.4.4.03.3	Other *Quercus* w/ pastures	NT
2.4.4	2.4.4.03	2.4.4.03.5	*Q. suber* and *Q. ilex* w/ pastures	NT
2.4.4	2.4.4.04	2.4.4.04.1	*Q. suber* w/ permanent cultures	NT
2.4.4	2.4.4.04	2.4.4.04.2	*Q. ilex* w/ permanent cultures	NT
2.4.4	2.4.4.04	2.4.4.04.3	Other *Quercus* w/ permanent cultures	NT
2.4.4	2.4.4.04	2.4.4.04.5	*Q. suber* and *Q. ilex* w/ permanent cultures	NT
Broadleaved forests (3.1.1)				
3.1.1	3.1.1.01	3.1.1.01.1	Pure *Q. suber* forests	NT
3.1.1	3.1.1.01	3.1.1.01.2	Pure *Q. ilex* forests	NT
3.1.1	3.1.1.01	3.1.1.01.3	Pure Other *Quercus* forests	NT
3.1.1	3.1.1.01	3.1.1.01.5	Pure *Eucalyptus* forests	NNT
3.1.1	3.1.1.01	3.1.1.01.6	Pure Invasive species forests	NNT
3.1.1	3.1.1.02	3.1.1.02.1	Dominated *Q. suber* forests	NT
3.1.1	3.1.1.02	3.1.1.02.2	Dominated *Q. ilex* forests	NT
3.1.1	3.1.1.02	3.1.1.02.3	Dominated Other *Quercus* forests	NT
3.1.1	3.1.1.02	3.1.1.02.5	Dominated *Eucalyptus* forests	NNT
3.1.1	3.1.1.02	3.1.1.02.6	Dominated Invasive species forests	NNT
Coniferous forests (3.1.2)				
3.1.2	3.1.2.01	3.1.2.01.1	Pure *Pinus pinaster* forests	NT
3.1.2	3.1.2.01	3.1.2.01.2	Pure *P. pinea* forests	NT
3.1.2	3.1.2.02	3.1.2.02.1	Dominated *P. pinaster* forests	NT
3.1.2	3.1.2.02	3.1.2.02.2	Dominated *P. pinea* forests	NT
Mixed forests (broadleaved w/ coniferous) (3.1.3)				
3.1.3	3.1.3.01	3.1.3.01.1	Mixed *Q. suber* w/ coniferous	NT
3.1.3	3.1.3.01	3.1.3.01.2	Mixed *Q. ilex* w/ coniferous	NT
3.1.3	3.1.3.01	3.1.3.01.3	Mixed *Quercus* w/ coniferous	NT
3.1.3	3.1.3.01	3.1.3.01.5	Mixed *Eucalyptus* w/ coniferous	NNT
3.1.3	3.1.3.01	3.1.3.01.6	Mixed Invasive species w/ coniferous	NNT
Open forests (3.2.4)				
3.2.4	3.2.4.01	3.2.4.01.1	Open *Q. suber* forests	NT

3.2.4	3.2.4.01	3.2.4.01.2	Open *Q. ilex* forests	NT
3.2.4	3.2.4.01	3.2.4.01.3	Open Other *Quercus* forests	NT
3.2.4	3.2.4.01	3.2.4.01.5	Open *Eucalyptus* forests	NNT
3.2.4	3.2.4.01	3.2.4.01.6	Open Invasive species forests	NNT
3.2.4	3.2.4.02	3.2.4.02.1	Open dominated *Q. suber* forests	NT
3.2.4	3.2.4.02	3.2.4.02.2	Open dominated *Q. ilex* forests	NT
3.2.4	3.2.4.02	3.2.4.02.3	Open dominated Other *Quercus* forests	NT
3.2.4	3.2.4.02	3.2.4.02.5	Open dominated *Eucalyptus* forests	NNT
3.2.4	3.2.4.02	3.2.4.02.6	Open dominated Invasive species forests	NNT
3.2.4	3.2.4.03	3.2.4.03.1	Open *P. pinaster* forests	NT
3.2.4	3.2.4.03	3.2.4.03.2	Open *P. pinea* forests	NT
3.2.4	3.2.4.04	3.2.4.04.1	Mixed *Pinus pinaster* forests w/ coniferous	NT
3.2.4	3.2.4.04	3.2.4.04.2	Mixed *P. pinea* forests w/ coniferous	NT
3.2.4	3.2.4.05	3.2.4.05.1	Open *Q. suber* forests w/ coniferous	NT
3.2.4	3.2.4.05	3.2.4.05.2	Open *Q. ilex* forests w/ coniferous	NT
3.2.4	3.2.4.05	3.2.4.05.3	Open Other *Quercus* forests w/ coniferous	NT
3.2.4	3.2.4.05	3.2.4.05.5	Open *Eucalyptus* forests w/ coniferous	NNT
3.2.4	3.2.4.05	3.2.4.05.6	Open Invasive species forests w/ coniferous	NNT

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Appendix C - Description of data sources of cultural ecosystem services

Four categories of cultural ecosystem services were considered, following the *Millennium Ecosystem Assessment* (MEA, 2005): Recreation and ecotourism, Aesthetic, Inspiration, and Cultural heritage. In total, our dataset comprised information from 305 data sources (64 for Portugal and 241 for Spain). The rational and details for each cultural ecosystem services data source are indicated below.

Recreation and ecotourism

Recreation and ecotourism services were assessed by means of two data sources: official tourism websites and nature routes.

Official tourism websites

Photographs used in tourism websites represent the "brands" of a place and aim to attract tourists to a region, so we assume that official tourism entities select those photographs that are more attractive for recreation and ecotourism.

We selected official tourism websites (avoiding those of private companies, particular accommodation services, among others), in order to homogenise the search across regions and to avoid repetition. Within each website, we focused on nature sections (e.g., natural areas, nature routes, outdoors sports) and selected at least 20 pictures dominated by identificable trees. At least one official tourism website was selected for each NUTS-2 region. For each tourism website, we counted the number of photographs dominated by non-native and native tree species.

For Spain were collected from the following official tourism websites:

http://www.andalucia.org/es/destinos/
http://www.turismodearagon.com/es/tipo-de-actividad/espacios-naturales#.WCxwJLLhAnQ
https://www.turismoasturias.es/descubre/naturaleza
http://www.illesbalears.es/esp/islasbaleares/naturaleza.jsp?SEC=NAT
http://www.turismodecantabria.com/disfrutala/recursos-naturales/
http://www.turismocastillalamancha.es/naturaleza/animales-y-bosques/
http://www.turismocastillayleon.com/
http://clicat.gencat.cat/clicat/AppJava//galeries.do?idColeccio=96ae8a85-868e-4c58-90da-f93f0703ef1a
http://www.turismodecastellon.com/ + http://www.valenciaturisme.org/es/
www.turismoextremadura.com/
http://www.turismo.gal/
https://lariojaturismo.com/comunidad/larioja

http://turismomadrid.es/es/naturaleza/10101-espacios-naturales.html
https://www.murciaturistica.es/es/naturaleza/
http://www.turismo.navarra.es/esp/organice-viaje/mapa.aspx?idioma=es
http://turismo.euskadi.eus/es/naturaleza/

Data for Portugal were collected from the following official tourism websites:
http://www.portoenorte.pt/
http://www.turismodocentro.pt/pt/
http://www.visitlisboa.com/
http://www.visitalentejo.pt/en/
http://www.turismodoalgarve.pt/home.html

Nature routes

People show preference for specific nature areas for recreation and ecotourism. There are websites where people can upload their geo-referenced routes (along with photographs taken along the route) to share them with the public. These routes and photographs allow evaluating whether people choose areas covered by non-native or native tree species or whether there is not a clear preference.

We obtained data from “Wikiloc” (available at http://www.wikiloc.com) because it covers routes worldwide. We selected routes more related to nature (including, hiking, cycling, walking), evenly distributed through the Iberian Peninsula. We then examined the photographs of the route. The selected routes were uploaded before the year 2016 and had a minimum of 10 photographs dominated by identifiable trees species. When this assumption was not verified, we considered another route nearby. Among the selected routes, we provided the number of photographs dominated by non-native or by native tree species. the sampling effort considered one valid route every 1600 km^2 of forested area in the country (ca. 120 routes for Spain and 25 routes for Portugal). for each route, we counted the number of pictures dominated by non-native tree species and native tree species in each wikiloc route.

Data for Spain were collected from the following nature routes on the “Wikiloc” website:
http://www.wikiloc.com/wikiloc/view.do?id=6076634
http://www.wikiloc.com/wikiloc/view.do?id=8485162
http://www.wikiloc.com/wikiloc/view.do?id=6587540
http://www.wikiloc.com/wikiloc/view.do?id=6937675
http://www.wikiloc.com/wikiloc/view.do?id=8101907
http://www.wikiloc.com/wikiloc/view.do?id=8160857
http://www.wikiloc.com/wikiloc/view.do?id=10697677
http://www.wikiloc.com/wikiloc/view.do?id=3809182
http://www.wikiloc.com/wikiloc/view.do?id=5872929
http://www.wikiloc.com/wikiloc/view.do?id=8864254
http://www.wikiloc.com/wikiloc/view.do?id=9529820
http://www.wikiloc.com/wikiloc/view.do?id=6944438
http://www.wikiloc.com/wikiloc/view.do?id=8536224
http://www.wikiloc.com/wikiloc/view.do?id=8222181
http://www.wikiloc.com/wikiloc/view.do?id=11628574
http://www.wikiloc.com/wikiloc/view.do?id=8122232
http://www.wikiloc.com/wikiloc/view.do?id=9039541
http://www.wikiloc.com/wikiloc/view.do?id=5651423

http://www.wikiloc.com/wikiloc/view.do?id=3672482
http://www.wikiloc.com/wikiloc/imgServer.do?id=5978526
http://www.wikiloc.com/wikiloc/view.do?id=5239568
http://www.wikiloc.com/wikiloc/view.do?id=4396160
http://www.wikiloc.com/wikiloc/view.do?id=11540373
http://www.wikiloc.com/wikiloc/view.do?id=11169153
http://www.wikiloc.com/wikiloc/view.do?id=4131784
http://www.wikiloc.com/wikiloc/view.do?id=4217101
http://www.wikiloc.com/wikiloc/view.do?id=10461232
http://www.wikiloc.com/wikiloc/view.do?id=10293376
http://www.wikiloc.com/wikiloc/view.do?id=10307304
http://www.wikiloc.com/wikiloc/view.do?id=7870794
http://www.wikiloc.com/wikiloc/view.do?id=9528357
http://www.wikiloc.com/wikiloc/view.do?id=8045980
http://www.wikiloc.com/wikiloc/view.do?id=7335878
http://www.wikiloc.com/wikiloc/view.do?id=4985956
http://www.wikiloc.com/wikiloc/view.do?id=6784335
http://www.wikiloc.com/wikiloc/imgServer.do?id=5518356
http://www.wikiloc.com/wikiloc/view.do?id=9680652
http://www.wikiloc.com/wikiloc/view.do?id=10258039
http://www.wikiloc.com/wikiloc/view.do?id=10933310
http://www.wikiloc.com/wikiloc/view.do?id=11196882
http://www.wikiloc.com/wikiloc/view.do?id=7276790
http://www.wikiloc.com/wikiloc/view.do?id=10840636
http://www.wikiloc.com/wikiloc/view.do?id=9252959
http://www.wikiloc.com/wikiloc/view.do?id=8322735
http://www.wikiloc.com/wikiloc/view.do?id=6553317
http://www.wikiloc.com/wikiloc/view.do?id=11238662
http://www.wikiloc.com/wikiloc/view.do?id=6089525
http://www.wikiloc.com/wikiloc/view.do?id=8133623
http://www.wikiloc.com/wikiloc/view.do?id=11629858
http://www.wikiloc.com/wikiloc/view.do?id=10919228
http://www.wikiloc.com/wikiloc/view.do?id=11576102
http://www.wikiloc.com/wikiloc/view.do?id=6976814
http://www.wikiloc.com/wikiloc/view.do?id=7595861
http://www.wikiloc.com/wikiloc/view.do?id=9588026
http://www.wikiloc.com/wikiloc/view.do?id=9579317
http://www.wikiloc.com/wikiloc/view.do?id=7209331
http://www.wikiloc.com/wikiloc/view.do?id=8233072
http://www.wikiloc.com/wikiloc/view.do?id=10565382
http://www.wikiloc.com/wikiloc/view.do?id=11186394
http://www.wikiloc.com/wikiloc/view.do?id=11182326
http://www.wikiloc.com/wikiloc/view.do?id=10220031
http://www.wikiloc.com/wikiloc/view.do?id=10097026
http://www.wikiloc.com/wikiloc/view.do?id=5712427
http://www.wikiloc.com/wikiloc/view.do?id=9568889
http://www.wikiloc.com/wikiloc/view.do?id=1240112
http://www.wikiloc.com/wikiloc/view.do?id=3573353
http://www.wikiloc.com/wikiloc/view.do?id=10591506
http://www.wikiloc.com/wikiloc/view.do?id=1798095
http://www.wikiloc.com/wikiloc/view.do?id=10484314
http://www.wikiloc.com/wikiloc/view.do?id=8964551
http://www.wikiloc.com/wikiloc/view.do?id=11508866
http://www.wikiloc.com/wikiloc/view.do?id=7251458
http://www.wikiloc.com/wikiloc/view.do?id=3641400
http://www.wikiloc.com/wikiloc/view.do?id=3580660
http://www.wikiloc.com/wikiloc/view.do?id=7919431
http://www.wikiloc.com/wikiloc/view.do?id=273685
http://www.wikiloc.com/wikiloc/view.do?id=289361
http://www.wikiloc.com/wikiloc/view.do?id=10015888
http://www.wikiloc.com/wikiloc/view.do?id=4644981
http://www.wikiloc.com/wikiloc/view.do?id=6844139
http://www.wikiloc.com/wikiloc/view.do?id=6264764
http://www.wikiloc.com/wikiloc/view.do?id=10303906
http://www.wikiloc.com/wikiloc/view.do?id=7493494
http://www.wikiloc.com/wikiloc/view.do?id=5499996
http://www.wikiloc.com/wikiloc/view.do?id=3495723
http://www.wikiloc.com/wikiloc/view.do?id=5238557
http://www.wikiloc.com/wikiloc/view.do?id=4530155
http://www.wikiloc.com/wikiloc/view.do?id=11335745
http://www.wikiloc.com/wikiloc/view.do?id=5222364
http://www.wikiloc.com/wikiloc/view.do?id=7568781
http://www.wikiloc.com/wikiloc/view.do?id=9097825
http://www.wikiloc.com/wikiloc/view.do?id=4646391
http://www.wikiloc.com/wikiloc/view.do?id=4595680
http://www.wikiloc.com/wikiloc/view.do?id=9570062
http://www.wikiloc.com/wikiloc/view.do?id=5681210
http://www.wikiloc.com/wikiloc/view.do?id=7948857
http://www.wikiloc.com/wikiloc/view.do?id=9679869
http://www.wikiloc.com/wikiloc/view.do?id=5416886
http://www.wikiloc.com/wikiloc/view.do?id=11310213
http://www.wikiloc.com/wikiloc/view.do?id=7655790
http://www.wikiloc.com/wikiloc/imgServer.do?id=5719676
http://www.wikiloc.com/wikiloc/view.do?id=8039508
http://www.wikiloc.com/wikiloc/view.do?id=9300880
http://www.wikiloc.com/wikiloc/view.do?id=10152184
http://www.wikiloc.com/wikiloc/view.do?id=9573404
http://www.wikiloc.com/wikiloc/view.do?id=4104880
http://www.wikiloc.com/wikiloc/view.do?id=3623808
http://www.wikiloc.com/wikiloc/view.do?id=8446663
http://www.wikiloc.com/wikiloc/view.do?id=3167707
http://www.wikiloc.com/wikiloc/view.do?id=10451352
http://www.wikiloc.com/wikiloc/view.do?id=8010212
http://www.wikiloc.com/wikiloc/view.do?id=4071624

http://www.wikiloc.com/wikiloc/view.do?id=11568196
http://www.wikiloc.com/wikiloc/view.do?id=8497789
http://www.wikiloc.com/wikiloc/view.do?id=9844601
http://www.wikiloc.com/wikiloc/view.do?id=9581364
http://www.wikiloc.com/wikiloc/view.do?id=8626481
http://www.wikiloc.com/wikiloc/view.do?id=3582200
http://www.wikiloc.com/wikiloc/view.do?id=7861273
http://www.wikiloc.com/wikiloc/view.do?id=11779822
http://www.wikiloc.com/wikiloc/view.do?id=11658804
http://www.wikiloc.com/wikiloc/view.do?id=11347063
http://www.wikiloc.com/wikiloc/view.do?id=8869044
http://www.wikiloc.com/wikiloc/view.do?id=11656304
http://www.wikiloc.com/wikiloc/view.do?id=7938605
http://es.wikiloc.com/wikiloc/view.do?id=8257559
http://www.wikiloc.com/wikiloc/view.do?id=10801825
http://www.wikiloc.com/wikiloc/view.do?id=8188436
http://www.wikiloc.com/wikiloc/view.do?id=6914454
http://www.wikiloc.com/wikiloc/view.do?id=7722717
http://www.wikiloc.com/wikiloc/view.do?id=3166601
http://www.wikiloc.com/wikiloc/view.do?id=6823841
http://www.wikiloc.com/wikiloc/view.do?id=9978423
http://www.wikiloc.com/wikiloc/view.do?id=4940829
http://www.wikiloc.com/wikiloc/view.do?id=5039022
http://www.wikiloc.com/wikiloc/view.do?id=11563128
http://www.wikiloc.com/wikiloc/view.do?id=6302429

Data from Portugal were collected from the following nature routes on the “Wikiloc” website:

http://www.wikiloc.com/wikiloc/view.do?id=11210163
http://pt.wikiloc.com/wikiloc/view.do?id=10438383
http://www.wikiloc.com/wikiloc/view.do?id=7995197
http://www.wikiloc.com/wikiloc/view.do?id=11422810
http://www.wikiloc.com/wikiloc/view.do?id=6561975
http://www.wikiloc.com/wikiloc/view.do?id=8640691
http://www.wikiloc.com/wikiloc/view.do?id=10712934
http://www.wikiloc.com/wikiloc/view.do?id=6619073
http://www.wikiloc.com/wikiloc/view.do?id=6191040
http://www.wikiloc.com/wikiloc/view.do?id=8393772
http://www.wikiloc.com/wikiloc/view.do?id=3523100
http://www.wikiloc.com/wikiloc/view.do?id=10712934
http://www.wikiloc.com/wikiloc/view.do?id=9438806
http://www.wikiloc.com/wikiloc/view.do?id=6331123
http://pt.wikiloc.com/wikiloc/view.do?id=5482178
http://www.wikiloc.com/wikiloc/view.do?id=9555543
http://www.wikiloc.com/wikiloc/view.do?id=9756071
http://www.wikiloc.com/wikiloc/view.do?id=967345
http://www.wikiloc.com/wikiloc/imgServer.do?id=3244001
http://www.wikiloc.com/wikiloc/view.do?id=10852779
http://www.wikiloc.com/wikiloc/imgServer.do?id=1024710
http://www.wikiloc.com/wikiloc/imgServer.do?id=5209808
http://www.wikiloc.com/wikiloc/view.do?id=3870514
http://www.wikiloc.com/wikiloc/view.do?id=1038627

Aesthetics

Aesthetic cultural ecosystem services were represented by two different data sources: catalogues of ornamental plant dealers and inventories of urban parks. The rationale and methods underlying each source are described below.

Catalogues of ornamental plant dealers

We assume that catalogues from ornamental plant dealers offer those ornamental species that are easily sold to the public. Therefore, tree species included in these catalogues can be appreciated for their aesthetic values. For each region in the Iberian Peninsula, we searched for online and printed catalogues of plant dealers of ornamental plants operating at each region (i.e., considering local plant dealers and avoiding international plant dealers). Our search was evenly distributed across Iberian NUTS-2 regions (1-3 catalogues per region, depending on its size and availability). For each catalogue, we counted the number of tree species (i.e., non-native and native tree species). different varieties, subspecies or cultivars within the same species were counted as one species. Fruit or forestry species were not considered, since they may not be representative of an aesthetic choice.

For Spain, we considered the following catalogues of plant dealers:

http://www.viveroszuaime.es
http://www.viverossevilla.com
http://viverosveron.com
http://www.viverosibanez.es
http://www.delsueve.com
http://viverossanchez.com
http://viverosferca.com
http://www.campogrande.es
http://viverosbarra.es
http://www.corma.es/intranet/products/search
http://www.urkiondo.com
http://viveroscanos.com
https://plantamus.com
http://www.coplant.es
http://www.viverosborrazas.com
http://viverosperica.com/es/catalogo-viveros-perica
http://www.viverosrucat.es
http://www.viverosametza.com/productos/listado_1.html
http://www.viverosbargues.com/arboles.html
http://www.alberolaviveros.com

For Portugal, we considered the following catalogues of plant dealers:

http://www.vcastromil.pt/
http://www.alfredomoreiradasilva.com/gestor/catalogo.php?cat=2
http://www.silvoplante.com
http://www.arborlusitania.com/
http://www.jardimdaceleste.com/
http://www.plantalivre.pt/
http://www.hortodorossio.com/
http://ecossistemasol.com/

Inventories of urban parks

Species planted in urban parks can be appreciated by people for their ornamental or aesthetic value.

Within each Iberian region we searched for inventories of urban parks for which the catalogue of tree species was available in the web or any other source (books, archives of municipalities, in-situ panels, among others). Information on trees planted in streets was not considered, as they may be selected based on different purposes (e.g., for the provision of shade). We used the maximum number of urban parks within each region for which information was available. For each urban park inventory, we counted the number of non-native tree species and native tree species. additional personal surveys (Portugal) and visits (Spain) to the urban parks were also done to complement the information.

Official inventories for urban parks from Spain were obtained from:

http://vivirlosparques.blob.core.windows.net/
http://www.granada.es/internet/arboles.nsf/lugc/32
http://xarxa018-1.edubcn.cat/fem/materialsaplec/arbresparcciutadella.pdf
http://www.madrid.es/UnidadesDescentralizadas/Educacion_Ambiental/ContenidosBasicos/Publicaciones/Retiro/SendaBot%C3%A1nicaRetiro11.pdf http://www.visitvalencia.com/es/Datos/IdiomaNeutral/PDF/parque_metropolitano.pdf

Official inventories for urban parks from Portugal were obtained from:

http://www.serralves.pt/pt/parque/
https://listasverdes.wordpress.com/2010/10/22/lista-especies-parque-da-cidade-porto/
http://www.icnf.pt/portal/florestas/gf/pgf/resource/doc/2010/mn-choupal/Anexo1%20-%20Flora.pdf
http://www.ipcb.pt/sites/default/files/upload/esa/files/agroforum/33.pdf
Soares, A. L. and C. Castel-Branco. 2007. As árvores da Cidade de Lisboa. Pages 289-334 in J. S. Silva, editor. Árvores e Florestas de Portugal, Vol. VII Floresta e sociedade - uma história em comum. Público/FLAD/LPN, Lisboa.
http://www.monumentos.pt/Site/APP_PagesUser/SIPA.aspx?id=2067
http://www.monumentos.pt/Site/APP_PagesUser/SIPA.aspx?id=17304
http://www.monumentos.pt/Site/APP_PagesUser/SIPA.aspx?id=28138

Cultural heritage

This cultural service was expressed by information on monumental tree species for Portugal and Spain, as described below.

Official list of monumental tree species

Many countries have an official catalogue of emblematic, monumental or singular tree species. These trees are appreciated symbols of the landscape and cultural heritage.

We considered official catalogues of emblematic/monumental/singular trees at the country level. For each catalogue, we counted the number of non-native and native tree species. Since the location of each monumental tree was provided in the lists of official catalogues, it was possible to gather this information for each region of the Iberian Peninsula.

Spanish data were obtained from the official catalogue of monumental tree species for each region:
http://www.juntadeandalucia.es/medioambiente/site/portalweb/menuitem.7e1cf46ddf59bb227a9ebe205510e1ca/?vgnextoid=d733e6ed46a28110VgnVCM1000000624e50aRCRD&vgnextchannel=d833a0b5f9ca5310VgnVCM2000000624e50aRCRD&lr=lang_es
https://www.clh.es/docs/PDFarboles_tomo2.pdf
https://www.larioja.org/npRioja/default/defaultpage.jsp?idtab=524420
http://www.madrid.org/cs/Satellite?blobcol=urldata&blobheader=application%2Fpdf&blobheadername1=Content-Disposition&blobheadervalue1=filename%3DCATALOGO_ESPECIES_2015.pdf&blobkey=id&blobtable=MungoBlobs&blobwhere=1352865924672&ssbinary=true

For Portugal, we used the official list for monumental available at: http://www.icnf.pt/portal/florestas/aip/aip-monum-pt. Since the location of each tree species was available, we derived the number of non-native native tree species for each NUTS-2 region.

Inspiration

Inspiration was assessed by means of artistic photographs.

Artistic nature photographs

Paintings, pottery, artistic photographs, show inspiring motifs. We can assume that an artist showing a photograph with nature motifs chooses these inspiring motifs from the nature of his/her environment.

We searched for websites on artistic photographs and contests of nature photographs. We selected the category of trees/forests and photographs indicating where they were taken. We used the maximum number of websites available for each country. Eligible websites compiled a selection of photographs from different authors (one-author personal websites were not valid). we selected photographs where non-native and native tree species could be identified and counted the number of photographs dominated by non-native or native tree species.

Data for Spain were obtained from the following websites:

www.miradanatural.es
www.fotonatura.org/galerias/flora
https://fotografiaperfecta.wordpress.com/2012/12/28/77-fotografos-de-naturaleza-que-deberias-conocer/
http://www.agefotostock.com/age/es/Photographers.aspx
http://migaleria-aefona.org/#/tematicas/paisajes/p_20
http://www.fonamad.org/galerias/

Data for Portugal were obtained from the following websites:

http://olhares.sapo.pt/
http://cinclusvouzela.com/galeria
http://www.ciencia20.up.pt/index.php?option=com_ciencia20galleries&view=system1&level1=13&level2=15&level3=59&Itemid=223
https://nationalgeographic.sapo.pt/a-sua-foto?limitstart=0
http://wildlifeportugal.pt/
http://fotografiaportugal.pt/

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Appendix D - Information on continuous predictors

Detailed information on the continuous predictors considered (Table S4.3).

Table S4.3. Type, description, data source, temporal and spatial characteristics of the continuous predictors considered in the structured meta-analysis, with respective references. Web links were last accessed in 22nd August 2018.

Description	Source	Temporal / spatial characteristics
***Land cover and nature protection** (Vilà and Pujadas, 2001; Vicente et al., 2016)*		
Forest areas (ForArea)		
Proportion of forested area by NUTS-2 (ha)	Portugal - IFN6 (http://www.icnf.pt/portal/florestas/ifn/resource/ficheiros/ifn/ifn6-res-prelimv1-1); Spain - IFN3 (http://www.mapama.gob.es/es/biodiversidad/servicios/banco-datos-naturaleza/informacion-disponible/ifn3.aspx)	Portugal - Photointerpretation of multitemporal land cover change for 1995, 2005, 2010 (minimum mapping area of 0,5 ha)/ Spain - Photointerpretation of multitemporal land cover change between 1997-2007 (1:50 000)
Protected areas (ProtArea)		
Proportion of national and international (RAMSAR and Natura 2000) protected and classified areas, incl., Special Protected Areas and Special Area of Conservation by NUTS-2 (ha)	Portugal - ICNF (http://www.icnf.pt/portal/naturaclas/cart); Spain - MAPAMA (http://www.mapama.gob.es/es/biodiversidad/servicios/banco-datos-naturaleza/informacion-disponible/cartografia_informacion_disp.aspx)	Areas for the year 2015 (1:50 000)
Socio-economy (Vilà and Pujadas, 2001; Krumm and Vítková, 2016)		
GDP		
Gross domestic product (GDP) at current market prices by NUTS-2 regions	Eurostat (http://ec.europa.eu/eurostat/web/regions/data/database)	Mean of the last 5 years (2009-2014) by NUTS-2
Tourism (Tour)		
Arrivals at tourist accommodation establishments by NUTS-2 regions	Eurostat (http://ec.europa.eu/eurostat/web/regions/data/database)	Mean of the last 5 years (2010-2015) by NUTS-2
Unemployment (Unemp)		
Total unemployment rate by NUTS-2 regions	Eurostat (http://ec.europa.eu/eurostat/web/regions/data/database)	Mean of the last 5 years (2010-2015) by NUTS-2

Human impact (Impact)		
Global Human Influence Index by NUTS-2 regions	NASA-SEDAC (http://sedac.ciesin.columbia.edu/data/set/wildareas-v2-human-influence-index-geographic)	Indicator of direct human influence on terrestrial ecosystems based on human settlement (population density, built-up areas), accessibility (roads, railroads, navigable rivers, coastline), landscape transformation (land-use/land-cover) and electric power infrastructure (night-time lights). Mean by NUTS-2 (30 arc-second grid cell size), values for 1995-2004
Human development index (HDI)		
EU regional human development index by NUTS-2 regions	European Commission Hardeman and Dijkstra (2014)	Regional values for the year 2012. The index is calculated grounded on several variables related with life expectancy, mortality, education, income, and employment
Human poverty index (HPI)		
EU regional poverty index by NUTS-2	European Commission Hardeman and Dijkstra (2014)	Regional values for the year 2012. The index is calculated grounded on several variables related with life expectancy and health, literacy, income, and unemployment
Human well-being *(OECD, 2013; Ghosh and Traverse, 2005; Vaz et al., 2017)*		
Income indicator (Income)		
Regional indicator of income by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Material conditions for well-being, measured through the household disposable income per capita (in real USD PPP), for 2010 (weighted by a scale 1 to 10)
Job indicator (Job)		
Regional indicator of job availability by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Material conditions for well-being, measured through both employment and unemployment rates (%), for 2010 (weighted by a scale 1 to 10)
Housing indicator (House)		
Regional indicator of housing by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Material conditions for well-being, measured through the ratio of the number of rooms per person (%), for 2010 (weighted by a scale 1 to 10)
Health indicator (Health)		
Regional indicator of health by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Quality of human life measured through both life expectancy at birth (years) and age adjusted mortality rate (per 1 000 people), for 2010 (weighted by a scale 1 to 10)
Education indicator (Educ)		
Regional indicator of education by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Quality of human life measured through the share of labour force with at least secondary education (%), for 2010 (weighted by a scale 1 to 10)
Environment indicator (Environ)		

Regional indicator of environment by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Quality of human life measured through the estimated average exposure to air pollution in PM2.5 (μg/m³), based on satellite imagery data, for 2010 (weighted by a scale 1 to 10)
Safety indicator (Safet)		
Regional indicator of human safety by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Quality of human life measured through the homicide rate (per 100 000 people), for 2010 (weighted by a scale 1 to 10)
Civic engagement (Civic)		
Regional indicator of civic engagement by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Quality of human life measured through the voter turnout (%), for 2010 (weighted by a scale 1 to 10)
Accessibility of services (Service)		
Regional indicator of accessibility to services by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Quality of human life measured through share of households with broadband access (%), for 2010 (weighted by a scale 1 to 10)
Community engagement (Comm)		
Regional indicator of community engagement by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Subjective well-being measured through the percentage of people who have friends or relatives to rely on in case of need, by 2010 (weighted by a scale 1 to 10)
Life satisfaction (Life)		
Regional indicator of life satisfaction by NUTS-2	OECD Regional well-being (https://www.oecdregionalwellbeing.org/)	Subjective well-being measured based on how people evaluate their life as a whole rather than their current feelings, by 2010 (scale 1 to 10)
Climate *(Gassó et al., 2009; Vicente et al., 2016)*		
Minimum temperature (MinTemp)		
Minimum temperature of the coldest month	Iberian Climate Atlas (http://opengis.uab.es/wms/iberia)	Mean values for NUTS-2 (250 m), values for 1971-2000
Range temperature (RangeTemp)		
Range of annual temperature	Iberian Climate Atlas (http://opengis.uab.es/wms/iberia)	Mean values for NUTS-2 (250 m), values for 1971-1999
Total precipitation (TotPrec)		
Total annual precipitation	Iberian Climate Atlas (http://opengis.uab.es/wms/iberia)	Mean values for NUTS-2 (250 m), values for 1971-2000
Precipitation driest month (DryPrec)		
Precipitation of the driest month	Iberian Climate Atlas (http://opengis.uab.es/wms/iberia)	Mean values for NUTS-2 (250 m), values for 1971-2001
Solar radiation (Rad)		

Annual solar radiation	Iberian Climate Atlas (http://opengis.uab.es/wms/iberia)	Mean values for NUTS-2 (250 m), values for 1971-2002

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Vilà, M., Pujadas, J., 2001. Land-use and socio-economic correlates of plant invasions in European and North African countries. Biol. Conserv. 100, 397-401.

Appendix E - Spearman correlation matrix for continuous predictors

This appendix shows the Spearman correlation matrix for the continuous predictors of land-cover and management, socio-economy, well-being and climate. Predictors with correlation values above 0.60 were not considered in subsequent analyses (Table S4.4).

Table S4.4. Spearman correlation values between the different continuous predictors. The selected (and uncorrelated, $p < 0.05$) predictors are highlighted with light-grey colour.

	For Area	Prot Area	GDP	Tour	Unemp	Impact	HDI	HPI	Safe	House	Life	Service	Civic	Educ	Job	Comm	Environ	Income	Health	Min Temp	Range Temp	Prec Tot	Prec Dry
ForArea																							
ProtArea	-0.478																						
GDP	0.480	-0.253																					
Tour	0.317	-0.149	0.259																				
Unemp	0.192	-0.341	-0.186	0.482																			
Impact	-0.194	0.263	0.316	0.109	-0.122																		
HDI	0.514	-0.290	0.842	0.054	-0.420	0.111																	
HPI	-0.385	0.251	-0.723	0.114	0.634	-0.037	-0.922																
Safe	0.568	-0.801	0.446	0.358	0.442	-0.097	0.359	-0.257															
House	0.224	-0.212	0.042	-0.468	-0.510	-0.540	0.333	-0.499	-0.032														
Life	0.709	-0.521	0.562	0.279	0.466	-0.091	0.495	-0.315	0.763	0.008													
Service	0.425	-0.241	0.882	0.414	0.121	0.519	0.640	-0.471	0.512	-0.280	0.666												
Civic	0.469	-0.663	0.499	0.429	0.540	-0.197	0.361	-0.191	0.810	-0.118	0.781	0.524											
Educ	0.504	-0.276	0.893	0.221	-0.130	0.234	0.808	-0.701	0.520	0.108	0.624	0.838	0.570										
Job	-0.373	0.481	-0.010	-0.488	-0.896	0.169	0.172	-0.362	-0.657	0.329	-0.642	-0.288	-0.698	-0.177									
Comm	0.615	-0.650	0.304	0.302	0.447	-0.027	0.377	-0.201	0.581	-0.097	0.762	0.473	0.610	0.455	-0.621								
Environ	0.123	0.299	-0.071	-0.308	-0.466	0.067	0.123	-0.140	-0.338	0.448	-0.294	-0.178	-0.484	0.042	0.346	-0.157							
Income	0.354	-0.053	0.919	0.030	-0.478	0.349	0.866	-0.802	0.194	0.201	0.332	0.729	0.216	0.843	0.270	0.167	0.152						
Health	0.636	-0.613	0.745	0.313	0.194	-0.128	0.655	-0.569	0.794	0.137	0.817	0.701	0.837	0.789	-0.467	0.567	-0.281	0.526					
MinTemp	-0.418	0.545	-0.538	-0.266	0.016	0.357	0.512	0.564	-0.678	-0.245	-0.511	-0.417	-0.651	-0.573	0.264	-0.294	0.203	-0.372	-0.816				
RangeTemp	-0.452	-0.122	-0.273	0.170	0.316	-0.281	0.425	0.433	-0.096	-0.280	-0.245	-0.292	0.166	-0.428	-0.101	-0.163	-0.412	-0.426	-0.160	-0.063			
PrecTot	0.402	-0.011	0.113	-0.216	-0.483	0.117	-0.402	-0.507	0.086	0.415	0.102	0.086	-0.264	0.308	0.199	0.209	0.507	0.298	0.091	-0.110	-0.809		
PrecDry	0.639	-0.207	0.529	-0.023	-0.287	-0.019	-0.651	-0.683	0.438	0.372	0.528	0.471	0.245	0.712	-0.056	0.380	0.277	0.582	0.594	-0.546	-0.761	0.772	
Rad	-0.396	0.019	-0.166	0.152	0.122	0.011	0.219	0.276	-0.263	-0.254	-0.331	-0.219	-0.096	-0.482	0.136	-0.238	-0.387	-0.247	-0.295	0.289	0.678	-0.684	-0.829

Appendix F - An indicator of non-native tree effects on cultural ecosystem services

Odds ratio

We proposed an indicator of effects of non-native tree species on cultural ecosystem services, based on the calculation of the odds ratio (OR). The OR can be computed from the following 2 x 2 contingency table (Table S4.5):

Table S4.5. Example of a contingency table for calculating the OR, based on the collection of information of non-native and native tree species, for cultural ecosystem services.

		Exposure	
		Exposure (non-native)	Non-exposure (native)
Outcome	Proportion of non-native and native tree species associated to a given cultural services	*A*	*B*
	Proportion of non-native and native tree species available at the nuts-2 region under analysis	*C*	*D*

In our case, we used Peto's method for OR computation, which is grounded on the following statistical procedures (Borenstein et al., 2008; Viechtbauer, 2010):

$$\Psi = exp\ (O\text{-}E/V)$$

$$O = A$$

$$E = (A+B)\ /\ (A+C)/n$$

$$V = (A+B)(C+D)(A+C)(B+D)\ /\ n\verb|^|2(n\text{-}1)$$

Where Ψ is Peto's odds ratio, n = A+B+C+D, and V is both weighting factor and variance for the difference between observed and expected A, O-E.

The OR was transformed as logOR, so that positive and negative values of logOR indicate that the contribution of non-native tree species to a data source in comparison to the contribution of native tree species, is respectively higher or lower than their proportion in the analysed region. Thus, logOR values higher or lower than 0 respectively indicate a positive or negative significant effect of non-native trees on a cultural service. LogOR equal to 0 indicate non-significant effects of non-native trees, i.e., that both non-native and native tree species were similarly frequent in the data sources. For each data source considered, measures of the number of non-native and native tree species related to the cultural ecosystem services (*outcome*), and measures reflecting non-native and native tree species expected by chance (*exposure*), are needed. The *exposure data* was often calculated considering the area covered by non-native and native tree SPECIES in each region/country under analysis of the Iberian Peninsula; yet, some data sources required different references (see details below). Since none of the two reference values (columns C and D) can be zero, if a given region has a non-native or native tree species cover = 0, or if non-native or native tree species are absent, that region was not been considered in the analysis (see details below).

For some data sources, we used values of different magnitudes. For example, for recreation and ecotourism, the observed non-native:native ratio was calculated from the number of non-native and native trees in the photographs, whereas the expected non-native:native ratio was obtained from the cover areas of non-native and native tree species in the region, resulting in unbalanced contingency tables. This means that the sum of values in a row of the contingency table (A and B) and the sum of values in the other row (C and D; Table S4.5) differed in their magnitude orders. Since Peto's method may fail in unbalanced contingency tables (Sweeting et al., 2004), we re-calculated the values used to calculate the expected ratio (C and D) dividing them by a constant that makes their sum equal to A + B while keeping the same non-native:native ratio of the original values.

Considerations for data collection

Data were searched at the regional level. Several Iberian participants were involved and responsible for the region that was the most familiar to them.

All data were gathered in the same template (excel) file. Each participant extracted the following information:

- Case study - each row was a case study (or observation). There were different observations for each region. The number of case studies/observations per region was proportional to the area covered by tree species/forest in each region;
- Participant - contained the initials of the name of the participants responsible for the information in each observation (row);
- Source - provided information on the source or reference of the information presented;
- Access date - referred to the date in which the information was gathered;
- Variable - description of the data source gathered by the participant;
- Category - the category of cultural ecosystem service considered in the observation;

- Country - the country for which the information relates to (Portugal or Spain);
- Region - the region for which the information relates to (NUTS-2 level);
- Province / City -information on the lower level spatial context of the information, whenever applied, i.e., the region/city for Portugal, and the state/province for Spain.

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Appendix G - Weighted log odds ratio of each data source of cultural ecosystem services

The weighted log odds ratio (logORw) was obtained under Peto's method, using both the DerSimonian-Laird random effects model, and the inverse variance with reciprocal of the opposite arm size zero-correction (Sweeting et al., 2004). Since both approaches showed similar results, the main manuscript only shows the results for the DerSimonian-Laird random effects model, with corresponding lower (Lower CI) and upper (Upper CI) 95% confidence intervals. The significance of logORw was obtained though non-parametric permutation tests (under 1000 iteractions). The significance of the heterogeneity of computed logOR for each data source is also shown and was obtained from a chi-squared test of the Q-statistic (QT). The chi-squared test was also applied to assess if the country (Portugal versus Spain) explained the variation across logORw (QM). Whenever a significant QM was obtained, we further computed the mean logORw for each separate country. The *rosenberg fail-safe numbers* generated for each data source are also shown: for each data source with a significant result, if the fail-safe numbers were larger than *5N + 10* (*N* is the number of observations), then we could be confident that the resulting logORw can be treated as reliable estimates of true effects (Table S4.6).

Table S4.6. Weighted logORw computed for each data source of cultural ecosystem services, with indication of the lower (Lower CI) and upper confidence (Upper CI) intervals, heterogeneity results (QT and QM), and Rosenberg fail-safe numbers. Statistical significance is highlighted with light grey cells at $p < 0.05$.

Peto's method (DerSimonian-Laird random)									**Inverse variance (zero correction model)**								**Rosemberg fail -safe numbers**	**5n+10**
Scale	***QM***	**p (*QM*)**	***QT***	**p *(QT)***	***logORw***	***p (logORw)***	**Lower CI**	**Upper CI**	***QM***	**p (*QM*)**	***QT***	**p *(QT)***	***logORw***	***p (logORw)***	**Lower CI**	**Upper CI**		
Tourism information																		
Iberia	9.593	0.170	**58.786**	**0.000**	**0.755**	**0.012**	0.169	1.340	7.658	0.085	**60.787**	**0.000**	0.837	0.066	-0.003	1.678	242	115
Portugal																	25.000	50
Spain																	235.000	110
Nature routes																		
Iberia	**52.901**	**0.000**	**283.512**	**0.000**	**-0.635**	**0.000**	-0.882	-0.388	**19.869**	**0.038**	**140.836**	**0.007**	**-0.860**	**0.002**	-1.297	-0.422	3207	815

Portugal			**58.218**	**0.000**	0.222	0.406	-0.311	0.756									4	130
Spain			**166.183**	**0.011**	**-0.948**	**0.000**	-1.186	-0.710									4522	695
Catalogues plant dealers																		
Iberia	**11.350**	**0.038**	**123.356**	**0.001**	**-0.353**	**0.008**	-0.603	-0.105	**19.942**	**0.000**	**110.283**	**0.000**	**-0.459**	**0.003**	**-0.707**	**-0.213**	279	150
Portugal			**4.005**	**0.676**	**0.472**	**0.036**	0.273	0.670									30	45
Spain			**105.390**	**0.000**	**-0.481**	**0.002**	-0.797	-0.166									269	110
Urban parks																		
Iberia	**24.152**	**0.002**	**127.247**	**0.000**	**-0.323**	**0.002**	**-0.518**	**-0.129**	**40.829**	**0.000**	**103.525**	**0.000**	**-0.478**	**0.001**	**-0.674**	**-0.282**	357	235
Portugal			15.717	0.073	0.221	0.172	-0.096	0.537									62	60
Spain			**85.494**	**0.000**	**-0.486**	**0.001**	**-0.693**	**-0.278**									492	185
Nature photographs																		
Iberia	1.186	0.404	14.253	0.219	-0.036	0.894	-0.694	0.623	5.023	0.170	10.281	0.505	0.454	0.522	-0.168	1.076	0	70
Portugal																		
Spain																		
Monumental trees																		
Iberia	23.613	0.272	**71.758**	**0.000**	**1.608**	**0.000**	0.917	2.299	11.768	0.088	**49.286**	**0.000**	**1.490**	**0.000**	0.687	2.293	508	115
Portugal																		
Spain																		

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Appendix H - Results of the structured meta-analysis

We report the heterogeneity explained by the regression model (QM) and its significance (p) based on a chi-square distribution with n-1 degree of freedom (Table S4.7).

Table S4.7. Results of the structured meta-analysis between the variation of log odds ratio (logOR) and the continuous predictors, and respective heterogeneity (QM). The regression slope, its standard error (SE) and its significance (p) are shown only when QM is significant ($p < 0.05$). Results are shown for each data source with a significant value for the total heterogeneity (QT) across observations. Statistical significance is highlighted with light grey at $p < 0.05$. See Table S4.4 for predictors description.

Predictor	**Model**	**df**	***QM***	**p (*QM*)**	**Slope**	**SE**	**p (regression)**
Websites of tourism information							
ForArea	Regression	1	0.0646	0.794			
	Residual	17	58.7842	< 0.0001			
	Total	18	58.8488				
ProtArea	Regression	1	0.1011	0.761			
	Residual	17	58.2589	< 0.0001			
	Total	18	58.36				
Tour	Regression	**1**	2.846	0.097			
	Residual	17	40.9515	0.0009			
	Total	18	43.7975				
HDI	Regression	1	0.0242	0.8870			
	Residual	17	56.3893	< 0.0001			
	Total	18	56.4135				
Impact	Regression	1	0.4606	0.484			
	Residual	17	57.0952	< 0.0001			
	Total	18	57.5558				
Life	Regression	1	1.3622	0.2670			
	Residual	17	51.9277	< 0.0001			
	Total	18	53.2899				
Jobs	Regression	1	5.0627	0.0320	-0.2849	0.1266	0.0320
	Residual	17	36.0332	0.0045			
	Total	18	41.0959				
House	Regression	1	0.0237	0.8740			
	Residual	17	58.0861	< 0.0001			
	Total	18	58.1098				
Environ	Regression	1	0.0001	0.9940			
	Residual	17	58.7838	< 0.0001			
	Total	18	58.7839				
MinTemp	Regression	1	0.2875	0.587			
	Residual	17	56.5668	< 0.0001			
	Total	18	56.8543				
PrecTot	Regression	1	0.4749	0.51			

	Residual	17	51.8934	< 0.0001			
	Total	18	52.3683				
Rad	Regression	1	0.0314	0.862			
	Residual	17	55.9972	< 0.0001			
	Total	18	56.0286				
Nature routes							
ForArea	Regression	1	13.5666	0.002	-0.0375	0.0102	0.004
	Residual	150	249.389	< 0.0001			
	Total	151	13.5666				
ProtArea	Regression	1	3.9919	0.054			
	Residual	150	276.0099	< 0.0001			
Tour	Regression	1	2.1993	0.16			
	Residual	150	277.9136	< 0.0001			
	Total	151	280.1129				
HDI	Regression	1	0.4241	0.5100			
	Residual	150	283.5114	< 0.0001			
	Total	151	283.9355				
Impact	Regression	1	0.6396	0.444			
	Residual	150	279.0967	< 0.0001			
	Total	151	279.7363				
Life	Regression	1	9.3897	0.0090	-0.5900	0.1926	0.0090
	Residual	150	240.9908	< 0.0001			
	Total	151	250.3805				
Jobs	Regression	1	25.2574	0.0010	-0.2680	0.0533	0.0010
	Residual	150	219.1746	< 0.0001			
	Total	151	244.4320				
House	Regression	1	1.6160	0.2040			
	Residual	150	280.5631	< 0.0001			
	Total	151	282.1791				
Environ	Regression	1	0.2228	0.6330			
	Residual	150	283.1394	< 0.0001			
	Total	151	283.3622				
MinTemp	Regression	1	1.9287	0.186			
	Residual	150	269.8981	< 0.0001			
	Total	151	269.8981				
PrecTot	Regression	1	5.9387	0.02	-0.0001	0	0.018
	Residual	150	251.8209	< 0.0001			
	Total	151	251.8209				
Rad	Regression	1	8.1782	0.009	0.0142	0.005	0.006
	Residual	150	240.8781	< 0.0001			
	Total	151	240.8781				
Catalogues of ornamental plant dealers							
ForArea	Regression	1	1.358	0.274			

	Residual	26	117.946	< 0.0001			
	Total	27	119.304				
ProtArea	Regression	1	0.823	0.350			
	Residual	26	120.086	< 0.0001			
	Total	26	120.909				
Tour	Regression	1	0.199	0.663			
	Residual	26	122.479	< 0.0001			
	Total	27	122.678				
HDI	Regression	1	4.283	0.037	-1.318	0.481	0.029
	Residual	26	117.8743	< 0.0001			
	Total	27	122.157				
Impact	Regression	1	0.986	0.339			
	Residual	26	117.703	< 0.0001			
	Total	27	118.689				
Life	Regression	1	5.961	0.025	-0.511	0.209	0.025
	Residual	26	99.775	< 0.0001			
	Total	27	105.736				
Jobs	Regression	1	0.366	0.595			
	Residual	26	120.100	< 0.0001			
	Total	27	120.466				
House	Regression	1	0.025	0.876			
	Residual	26	123.283	< 0.0001			
	Total	27	123.308				
Environ	Regression	1	0.489	0.525			
	Residual	26	122.892	< 0.0001			
	Total	27	123.381				
MinTemp	Regression	1	4.299	0.043	0.114	0.055	0.043
	Residual	26	104.429	< 0.0001			
	Total	27	108.728				
PrecTot	Regression	1	0.745	0.412			
	Residual	26	119.997	< 0.0001			
	Total	27	120.742				
Rad	Regression	1	0.872	0.339			
	Residual	26	118.921	< 0.0001			
	Total	27	119.793				
Inventories of urban parks							
ForArea	Regression	1	2.911	0.097			
	Residual	43	120.067	< 0.0001			
	Total	44	122.978				
ProtArea	Regression	1	3.168	0.116			
	Residual	43	115.015	< 0.0001			
	Total	44	118.183				
Tour	Regression	1	11.282	0.002	-0.0001	0.000	0.002

	Residual	43	99.628	< 0.0001			
	Total	44	111.910				
HDI	Regression	1	0.160	0.702			
	Residual	43	127.032	< 0.0001			
	Total	44	127.192				
Impact	Regression	1	1.102	0.315			
	Residual	43	122.953	< 0.0001			
	Total	44	124.055				
Life	Regression	1	4.721	0.043	-0.384	0.177	0.043
	Residual	43	114.477	< 0.0001			
	Total	44	119.198				
Jobs	Regression	1	7.497	0.014	-0.132	0.048	0.014
	Residual	43	104.540	< 0.0001			
	Total	44	112.037				
House	Regression	1	0.001	0.965			
	Residual	43	126.748	< 0.0001			
	Total	44	126.749				
Environ	Regression	1	4.737	0.0640			
	Residual	43	110.609	< 0.0001			
	Total	44	115.346				
MinTemp	Regression	1	5.703	0.023	0.114	0.047	0.023
	Residual	43	115.698	< 0.0001			
	Total	44	121.401				
PrecTot	Regression	1	2.482	0.123			
	Residual	43	116.402	< 0.0001			
	Total	44	118.884				
Rad	Regression	1	0.407	0.506			
	Residual	43	123.986	< 0.0001			
	Total	44	124.393				
Monumental tree species							
ForArea	Regression	1	0.9519	0.342			
	Residual	16	71.743	< 0.0001			
	Total	17	72.6949				
ProtArea	Regression	1	0.496	0.512			
	Residual	16	68.2367	< 0.0001			
	Total	17	68.7327				
Tour	Regression	1	0.0589	0.8			
	Residual	16	56.9289	< 0.0001			
	Total	17	56.9878				
HDI	Regression	1	0.3235	0.5840			
	Residual	16	53.9391	< 0.0001			
	Total	17	54.2626				
Impact	Regression	1	2.9778	0.12			
	Residual	16	65.0429	< 0.0001			

	Total	17	68.0207	
Life	Regression	1	1.7745	0.2080
	Residual	16	69.4142	< 0.0001
	Total	17	71.1887	
Jobs	Regression	1	1.2891	0.2690
	Residual	16	64.6752	< 0.0001
	Total	17	65.9643	
House	Regression	1	0.2563	0.6370
	Residual	16	62.2267	< 0.0001
	Total	17	62.4830	
Environ	Regression	1	4.0825	0.0610
	Residual	16	63.5103	< 0.0001
	Total	17	67.5928	
MinTemp	Regression	**1**	5.3012	0.0510
	Residual	16	37.3309	< 0.0001
	Total	17	42.6321	
PrecTot	Regression	1	0.4654	0.499
	Residual	16	59.8927	< 0.0001
	Total	17	60.3581	
Rad	Regression	1	0.0885	0.778
	Residual	16	71.3073	< 0.0001
	Total	17	71.3958	

CHAPTER 5. REVIEW OF REMOTE SENSING IN INVASION SCIENCE

DISCLAIMER

This chapter is an original contribution of this thesis, which has been published as a review paper in journal *Science of the Total Environment* under the title "*Managing plant invasions through the lens of remote sensing: recent progress and the way forward*". The paper had the contribution of the following authors:

Ana Sofia Vaz[a,b,*], Domingo Alcaraz-Segura[c,d,e], João C. Campos[a], Joana R. Vicente[a,i], João P. Honrado[a,b]

[a]Research Network in Biodiversity and Evolutionary Biology, Research Centre in Biodiversity and Genetic Resources (InBIO-CIBIO), Campus Agrário de Vairão, Rua Padre Armando Quintas, PT4485-661 Vairão, Portugal;

[b]Faculdade de Ciências, Universidade do Porto, Rua do Campo Alegre, s/n, PT4169-007 Porto, Portugal;

[c]Departamento de Botánica, Facultad de Ciencias, Av. Fuentenueva, Universidad de Granada, 18071, Granada, Spain;

[d]iecolab. Interuniversitary Institute for Earth System Research (IISTA) - Universidad de Granada, Av. del Mediterráneo, 18006, Granada, Spain;

[e]Andalusian Center for the Assessment and Monitoring of Global Change (CAESCG) - Universidad de Almería, Crta. San Urbano, 04120, Almería, Spain;

[i]Laboratory of Applied Ecology, CITAB − Centre for the Research and Technology of Agro-Environment and Biological Sciences, University of Trás-os-Montes e Alto Douro, Vila Real, Portugal.

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ABSTRACT

Biological invasions are a challenging driver of global environmental change and a fingerprint of the Anthropocene. Remote sensing has gradually become a fundamental tool for understanding invasion patterns, processes and impacts. Nevertheless, a quantitative overview of the progress and extent of remote sensing applications to the management of plant invasions is lacking. This overview is particularly necessary to support the development of more operational frameworks based on remote sensing that can effectively improve the management of invasions. Here, we evaluate and discuss the progress, current state and future opportunities of remote sensing for the research and management of plant invasions. Supported on a systematic literature review, our study shows that, since the 1970s, remote sensing was mainly used to map and identify invasive plants, evolving, around the mid-2000s, towards a tool for assessing invasion impacts. Although remote sensing studies often focus on detecting plant invaders at advanced invasion stages, they can also contribute to the prediction of early invasion stages and to the assessment of their impacts. Despite the growing awareness of technical limitations, remote sensing offers many opportunities to further improve the management of plant invasions. These opportunities relate to the capacity of remote sensing to: (a) detect and evaluate the extent of invasions, assisting on any management option aiming at mitigating plant invasions and their impacts; (b) consider modelling frameworks that anticipate future invasions, supporting the prevention and eradication at early invasion stages and protecting ecosystems and the services they provide; and (c) monitor changes in invasion dominance, as well as the resulting impacts, supporting mitigation, restoration and adaptation actions. Finally, we discuss the way forward to make remote sensing more effective in the scope of invasion management, considering current and future Earth observation missions.

Keywords: Earth observation; Ecosystem service; Invasion effect; Monitoring; Non-native plant

5.1. INTRODUCTION

The globalisation of trade and other human activities have caused the spread of non-native plant species worldwide (Meyerson and Mooney, 2007; Kueffer, 2017). Often, non-native plants have been introduced to improve natural resources, to provide ecosystem services or to minimise ecosystem disservices (Simberloff et al. 2013; Vaz et al., 2017a). However, non-native species may become invasive (*sensu* Richardson et al., 2011), i.e. spreading outside their native range, becoming abundant and leading to impacts on biodiversity, ecosystem functioning and human well-being (e.g. culture, health, economy; Simberloff et al., 2013; Vaz et al., 2018). Thus, there is an increasing commitment from researchers, managers and policy-makers to manage plant invasions and their impacts (e.g. EU Regulation 1143/2014; Buchadas et al., 2017; Courtois et al. 2018; Vaz et al., 2017b). For long, such efforts have tried to eradicate plant species and contain effects of the invasion process with difficult and costly management options (Meyerson and Mooney, 2007). More recently, pragmatic management options aim also to prevent and early detect new invasions (Juanes, 2018; Simberloff et al., 2013), as well as to adapt to their impacts (Kueffer, 2017; Vaz et al., 2017a). This raises the need to devise and implement time- and cost-efficient options that can provide decision-makers and other stakeholders with the best information for management solutions.

Remote sensing, i.e. the process of remotely capturing information about the Earth, has become increasingly important for environmental conservation and ecological monitoring (Kwork, 2018; Murray et al., 2018; Rose et al., 2014; Vaz et al., 2015), including the management of invasive species (Dvořák et al. 2015; Juanes, 2018; Manfreda et al., 2018; Müllerová et al., 2013, 2017a, 2017b). Over the last decades, remote sensing has contributed to improve understanding on the drivers, processes and effects of plant invasions. For instance, it has been used in the identification of plant invasions and invaded ecosystems (e.g. Alvarez-Taboada, 2017; Peerbhay et al., 2016; Müllerová et al., 2017b), in the prediction of the potential distribution of invasive species (He et al., 2011; López and Stokes, 2016; Rocchini et al., 2015), and in comprehending landscape invasibility and associated ecological impacts (Hellmann et al., 2017; Truong et al., 2017). Recently, remote sensing has also shown high potential to assess impacts on ecosystem functional attributes, properties and services (Andrew et al., 2014; Dzikiti et al., 2016; Hellmann et al., 2017; Niphadkar and Nagendra, 2016; Pettorelli et al., 2017; West et al., 2016).

Given the growth of remote sensing, a review of its progress and extent of application in the research and management of plant invasions is needed to advance invasion science. Previous studies have partially reviewed applications on the remote mapping of invasive plants based on spectral, textural or phenological analyses (e.g. Bradley, 2014; He et al., 2011); discussed remote sensing efforts towards plant invasions from different spatial, temporal and spectral perspectives (e.g. Huang and Asner, 2009; Müllerová et al., 2017a); compared remote sensing approaches for invasion monitoring (e.g. Dvořák et al., 2015; Müllerová et al., 2017b); or approached future applications of remote sensing in invasions (e.g. Niphadkar and Nagendra, 2016), targeting particular invasive species (e.g. Thamaga and Dube, 2018) or management strategies (e.g. Juanes et al., 2018). Analysing the development, current situation and challenges of remote sensing applications to plant invasions could help on guiding research opportunities, developing future remote sensing strategies and delineating practical management solutions.

In this study, we evaluate and discuss the progress, current state and opportunities of remote sensing applications in the research and management of plant invasions. Specifically, we aim to: (a) examine how remote sensing applications on plant invasions have evolved through time; (b) identify current data sources and remote sensing contributions for managing plant invaders and their impacts at distinct invasion stages; and (c) identify current challenges of remote sensing and discuss future opportunities in the scope of plant invasions. The ideas presented in this study are supported by a systematic literature review and by the authors' experience with the application of remote sensing to plant invasions and to other social-ecological challenges. Our rationale is grounded on a management framework that considers the distinct stages of the invasion process, the potential impacts of plant invaders on ecosystem services, and the contributions of remote sensing for the management of invasions and of their impacts. Finally, we discuss possible ways forward in invasion science, considering current and future remote sensing methods and missions.

5.2. METHODS

5.2.1. Literature search

A literature search on non-native/invasive plants was conducted in ISI Web of Science (at: http:// webofknowledge.com/) and Scopus (at: https://www.scopus.com/) core search engines. Keyword selection followed a Population-Intervention-Comparison-Outcome (PICO) strategy (Higgins and Green, 2011), in which “invasive plants” were defined as Population, “remote sensing” as Intervention, and “management” as Outcome. The selection of keywords was achieved under a participatory approach, with a team of researchers experienced in remote sensing (DAS, JCC), biological invasions (ASV, JRV) and plant ecology (JPH), and was further completed with a revision of keywords from a list of reference papers. The final list of keywords included the most common and unambiguous words, in order to reach the largest number of records published on the subject. Additional searches on the subject of plant invasions were conducted in 92 different publication sources, specialised in remote sensing (including journals, proceedings and books; see Appendix A for details on keyword selection and search procedure). The time span of our search was from 1950 to 2016, corresponding to the period when the systematic study of invasive species began (Richardson and Pyšek, 2008). All searches were conducted in June-July of 2017 and were updated in March of 2018.

After the elimination of duplicates, the records retrieved by all searches were combined (total number of records, n = 704) using EndNote 7.4 (Thomson Reuters, 2013). The reliability of our search was evaluated by comparing the first 50 records retrieved by Google Scholar (using the main keywords “plant invasion” AND “remote sensing” AND “management”) against the combined database (following Higgins and Green, 2011). All records on the topic found on Google Scholar, were already part of the combined database. Finally, non-relevant records were discarded, e.g. those mentioning the word “management”, which did not focus on management themselves. Exclusion criteria were applied by checking the title and keywords of each individual record (see Appendix B for details on inclusion/exclusion criteria), resulting in a final database of 289 records (Figure 5.1).

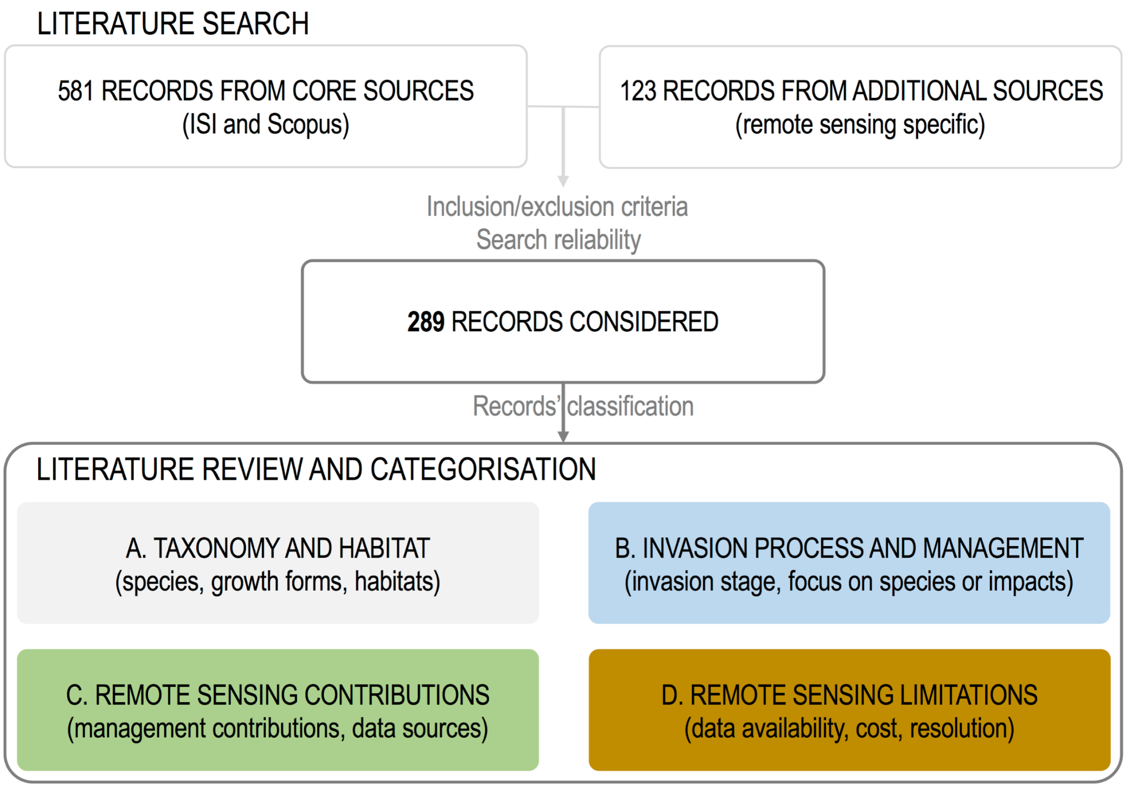

Figure 5.1. Analytical framework considered for reviewing remote sensing applications in the scope of plant invasions. We first searched for records focused on remote sensing and plant invasions in ISI Web of Science and Scopus (core sources) and in publication sources specialised in remote sensing (additional sources; time period of the search: 1950-2016). We then applied criteria to exclude irrelevant records. Finally, records were reviewed for: (A) taxonomic and habitat characterisation; (B) invasion process and management focus (on the species and/or their impacts); (C) types of remote sensing contributions (including main sources of remote sensing data); and (D) remote sensing limitations (see also Table 5.1).

5.2.2. Literature review

In order to assess the main contributions, limitations and opportunities of remote sensing to the management of plant invasions (objectives 2 and 3), the full text of each individual record from the final database (n = 289) was reviewed and categorised according to four main categories shown in Table 5.1 (A-D). First, we made a taxonomic and habitat characterisation of the records to identify the invasive plant species and respective growth form, and the habitat type under invasion (Table 5.1-A). Then, we reviewed the invasion process and management focus of each record to identify which stages of the invasion process were targeted and whether management focused on the invasive species and/or the impacts of the invader (Table 5.1-B). We further identified the contributions of remote sensing for managing invasions, including the main types of data sources used for that purpose (Table 5.1-C). Finally, we reviewed the main limitations of remote sensing for managing plant invasions (see

Table 5.1-D). For illustrative purposes, we gathered information from each record on the main characteristics of remote sensing sources (e.g. type of satellite sensor), data (e.g. hyperspectral, multispectral or radar information) and products (e.g. related to water, soil, vegetation properties; see Appendix C).

Table 5.1. Questions and categories used for categorising the records retrieved from our search. First, we obtained a taxonomic and habitat characterisation of the record (A); then, we reviewed the invasion process and management focus of each record (B) and identified the main sources and contributions of remote sensing (C); finally, we identified the main constraints of remote sensing for the management of plant invasion (D).

Questions/categories	Description
A. Taxonomic and habitat characterisation	
A1. Which species and growth forms have been mostly managed using remote sensing?	
Species	The name of the species referred by the record afterwards adapted from The Plant List (at http://www.theplantlist.org/)
Growth form	The growth form of the species: herbaceous, shrubs, trees, succulents or ferns
A2. Which invaded habitats have been mostly targeted by remote sensing?	
Habitat	Targeted habitats classified based on the habitat classification scheme from IUCN (at: http://www.iucnredlist.org/)
B. Invasion process and management focus	
B1. Which stages of the invasion process are studied*? (Vaz et al., 2017b)*	
Introduction	Pathways of species introduction from one geographical region to another
Establishment	Determinants of success of species establishment
Expansion	Patterns and mechanisms of species expansion
Dominance	Patterns and processes of invaders that became dominant in the invaded area
B2. What is the main focus of remote sensing approaches? *(Pau and Dee, 2016)*	
Invasive species	Management focuses on the species itself
Invasion impacts	Management focuses on the changes induced by the species (e.g. on soil, carbon and water cycles, fire regime, land cover)
C. Remote sensing sources and types of contributions	
C1. Which sources of remote sensing data have been contributing the most? *(Toth and Józków, 2016)*	
Airborne	Data is retrieved mostly by manned aerial vehicles (e.g. airplanes, helicopters)
UAV	Data is retrieved mostly by unmanned aerial vehicles (e.g. drones)
Field measurements	Data is collected on the ground, using field remote sensing instruments (e.g. field spectrometer, tractors, towers or other equipment-mounted sensors on the ground)
Satellite	Data is retrieved mostly by satellite instruments
C2. What are the main contributions of remote sensing for managing invasions? *(He et al., 2011)*	
Detect	Focused on the detectability of target invasive species or invaded habitats
Predict	Predicting and anticipating which species are more likely to invade (invasiveness), or which areas are more likely to be invaded (invasibility)
Assess	Evaluate recent or on-going impacts on the invaded area
D. Remote sensing limitations	
D1. Which are the main limitations of remote sensing for managing invasions? *(Turner et al., 2015)*	

Data availability	Data necessary to obtain clarified information is still not available
Data cost	Costs of data acquisition or reproducibility are expensive
Field validation	Results can only be seen as accurate if proper field calibration or validation is done
Radiometric resolution	The radiometric resolution of available data is insufficient to get accurate results
Spatial resolution	The spatial resolution of available data is insufficient to obtain accurate results
Spectral resolution	The spectral resolution of available data is insufficient to obtain accurate results
Temporal resolution	The temporal resolution of available data is insufficient to obtain accurate results
Technical constraints	Technical and methodological approaches are the major limitations (e.g. data storage, computational power, processing time)

5.2.3. Data analysis

In order to examine how remote sensing applications on plant invasions have evolved through time (objective 1), we used descriptive statistics to show temporal trends of published records. Specifically, the total number of published records per year was plotted as smoothing curves (averages for 2-year time periods) between 1977 (first record in our search) and 2016 (see section 5.3). Then, to assess how remote sensing has been considered in invasion management (objective 2), the proportion of records of each source and type of contribution from remote sensing was represented across stages of the invasion process. This was done by using column plots/bar charts in relation to the total number of published records per year (see section 5.4). Furthermore, to explore similarities among classifications, we applied a Principal Component Analysis (PCA) to the categories of "invasion process", "management focus", "remote sensing sources", and "remote sensing contributions" (Figure 5.1, Table 5.1), following Buchadas et al. (2017). Finally, remote sensing limitations (objective 3) were represented through radar plots for different time-periods, using a logarithmic scale to allow comparisons across years (see section 5.5). All statistical procedures were conducted using Statistica v13 (StatCorp, 2013).

5.3. EVOLUTION OF REMOTE SENSING APPLICATIONS TO PLANT INVASIONS

Invasion science and remote sensing are recently established disciplines. The first has seen its beginnings in the second-half of the 20th century (Richardson and Pyšek, 2008), after the publication of *"The Ecology of Invasions by Animals and Plants"* by Charles Elton (Elton, 1958). Remote sensing seems to have followed the progress of engineering and technological sciences, which had a consistent presence in the literature of invasions since the last couple of decades (Vaz et al., 2017b). Our bibliographic search highlights that, despite the start of invasion science in the 1960s and its evident increase in the 1990s, remote sensing applications in invasion literature only emerged in the late 1970s, showing a steady presence in the early 2000s, and a rapid increase in the mid-2000s (Fig. 5.2a).

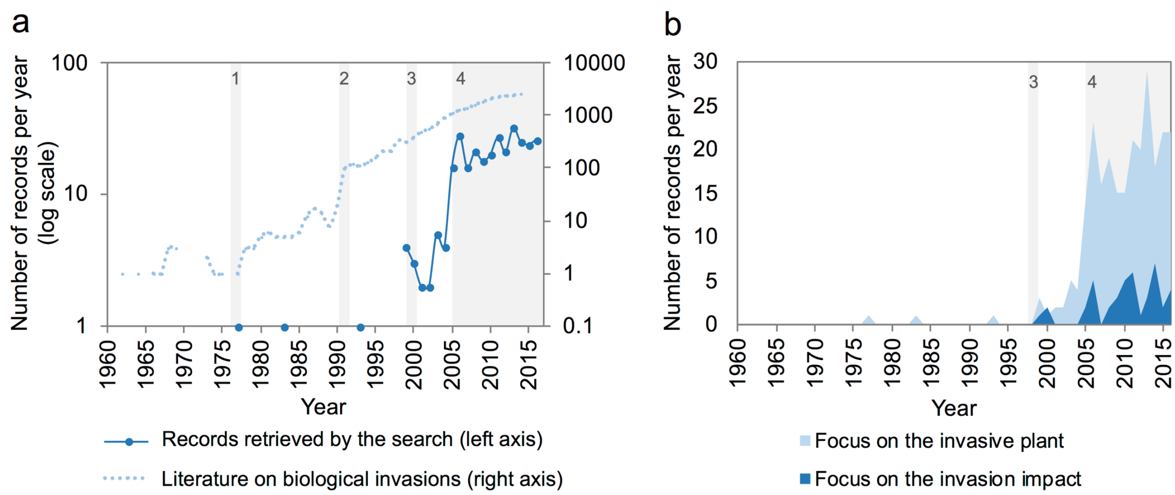

Figure 5.2. Temporal trends in the number of published records: (a) resulting from our search on plant invasions and remote sensing, and from a search on biological invasions for comparison (retrieved from Vaz et al., 2017b); (b) based on whether the management focus was on invasive plant species or on invasion impacts. Time periods highlighted in light grey are discussed in detail throughout the text and refer to: emergence of remote sensing in plant invasions in 1977 (period 1); steep increase of publications on invasions since 1990 (period 2) and using remote sensing since 2000 (period 3); and the bloom of remote sensing applications in plant invasions since 2005 (period 4).

Our analyses also demonstrate that there has been a sporadic interest in the application of remote sensing to plant invasions since the 1970s (Figure 5.2a, time period 1). These early studies (e.g. Capehart et al., 1977; Musick, 1983) attempted, though mostly unsuccessfully, to map invaded habitats through satellite imagery. Conservation and environmental problems were increasingly recognised in the 1970s and 1980s, with the emergence of environmental management, biodiversity conservation and restoration ecology, in which the spread of non-native plants also became an issue of concern (Zhang et al., 2010). During the 1970s and 1980s, remote sensing reached important milestones with the launching of the first multi- and hyperspectral satellites (Kwok, 2018). The Landsat project, in 1972, started the era of multispectral imagery, motivating intensive developments during the following 20 years, e.g. with the launch of four Landsat satellites between 1972-1984 and the first SPOT satellite in 1986 (Filchev, 2014). This allowed the acquirement of data applicable in conservation and ecology studies with global coverage and high spatial, temporal and spectral resolutions. However, the first remote sensing studies were mostly applied to agriculture monitoring and forest mapping, without focusing on invasions. Additionally, remote sensing applications prioritised research on natural hazards or disasters (Pettorelli et al., 2014; Rose et al., 2014).

The acceleration of publications in invasion science in the 1990s (Figure 5.2a, time period 2), mostly as a consequence of the international SCOPE program on biological invasions in 1982 (Davis, 2006), was not followed by an increasing use of remote sensing in this field. Instead, remote sensing seems to have only showed a steady prevalence in the beginning of the 2000s (Figure 5.2, time period 3), and an evident growth since the middle of this decade (Figure 5.2, time period 4). This increase, particularly since the mid-2000s, might reflect a growing access to remote sensing-based products, such as those from STRM (*Shuttle Radar Topography Mission*) and TRMM (*Tropical Rainfall Measuring Mission*) missions, which respectively provided elevation and precipitation data, since the early 2000s. The free access to the complete Landsat archive since 2009 (e.g. Skidmore and Pettorelli, 2015), as well as to online services and software that rely on remote sensing imagery in a map-like format (e.g. Google Earth, released in 2005; Turner et al., 2015) also contributed to the rise of remote sensing.

Plant invasions constituted a new topic for scientific research and publication in the 2000s, particularly in applied and restoration ecology (Hobbs and Richardson, 2011). These subjects also ensured the integration of plant invasions in wider interdisciplinarity perspectives (e.g. *Intergovernmental Platform on Biodiversity and Ecosystem Services* - IPBES), for which remote sensing constituted an increasingly promising tool (Pettorelli et al., 2014, 2017). Thus, it is not surprising to note that focus on invasion impacts gained higher prominence since the mid-2000s (Figure 5.2b). It was during this time that concerns on ecosystem services emerged (e.g. through the *Millennium Ecosystem Assessment*) and the *Global Invasive Species Programme* was established to foster cooperation among natural sciences, technological applications and decision-making (Hui and Richardson, 2017). During the 2000s, remote sensing seems to have moved from an issue concerning the mapping and identification of invasive plants, towards a tool to assess invasion impacts (Figure 5.2b), namely on soil nutrients (e.g. as illustrated in Hellman et al., 2017), carbon (Asner et al., 2010) and water cycles (e.g. Dzikiti et al., 2016; Espinar et al., 2015), or fire regime (e.g. Ellsworth et al., 2014).

Furthermore, as more data from remote sensing became available, concern on invasions also resulted in a greater availability of data on plant invaders (e.g. as demonstrated by the recent *Global Register of Introduced and Invasive Species*; Pagad et al., 2018), allowing for the application of more quantitative analyses and for the rising of predictive modelling and geographic information systems in invasion research (Buchadas et al., 2017; Rocchini et al., 2015). The slower uptake of remote sensing in invasion research until the 2000s may suggest that, until then, there was a focus on the use of technological solutions for managing plant invaders (Vaz et al., 2017b). A growing interest in complex mathematical models for elucidating aspects of invasion dynamics can be verified during this period (e.g. habitat suitability models; Buchadas et al., 2017; Guisan and Zimmermann, 2000). However, despite the available technology, management interventions were not completely successful (Simberloff, 2001), leading to the search for additional solutions later in the decade, in which remote sensing could succeed (e.g. He et al., 2015; Rocchini et al., 2015). In fact, successful advances in remote sensing during the last years seem to be characterised by an increasing interdisciplinarity (Skidmore and Pettorelli, 2015). This is demonstrated by the recent integration of sciences which aid in understanding and managing invasions, as exemplified by those focused on environmental DNA (Bush et al., 2017) and functional tracers (Hellmann et al., 2017), as well as, on social media and data analytics (Kissling et al. 2017; Mathieu and Aubrecht, 2018; Toth and Józków, 2016).

In sum, the evolution of remote sensing in the management of plant invasions seems to result from general technological advances and developments in the history of invasion science. The following highpoints can be emphasised: (1) remote sensing as a new tool in ecology, conservation and environmental management since the 1970s; (2) plant invasions, in line with other global challenges (e.g. climate and land use changes), as a preeminent topic for applied research and political debate in the 1990s; (3) feedbacks between scientific and political interests, with increasing availability of remote sensing products and data on plant invasions since the 2000s; and (4) increasing cross-collaboration together with higher availability of technological solutions and remote sensing products and services since the mid-2000s, when solid grounding and interdisciplinarity in invasion research also occurred.

5.4. Remote sensing applications in the management of plant invasions

5.4.1. Targeted species and habitat types

Our review shows that 49% of remote sensing applications deal with the management of herbaceous invaders, followed by trees (44%), shrubs (3%), ferns, and succulents (1.5% each; see Appendix D for details). Early research focused on tree species; however, it was gradually replaced by a focus on smaller plants. The earliest record from our search (Capehart et al., 1977) focused on the mapping of invaded habitats by the tree *Melaleuca quinquenervia* in the USA. Since 2000, a diversification of targeted tree species was observed, including a large number of studies on *Prosopis glandulosa* and *Tamarix ramosissima* (comprising ca. 10% of all studies; see Appendix D for species proportions). During this period, a growing interest on the application of remote sensing to the management of herbaceous invaders was also observed, namely on *Spartina alterniflora*, *Eichhornia crassipes* or *Phragmites australis* (Appendix D).

A wide range of invaded habitat types has also been targeted, with forests (including forest wetlands and plantations) receiving a higher focus (33% of records). Shrublands (15%), grasslands (13%), arable lands and pasturelands (6% each) were also among the most targeted habitat types, followed by a smaller number of studies in estuaries, coastal dunes, freshwater systems, and marshlands (Appendix D). The potential of invasion and respective magnitude of impacts are amongst the main motivations driving the selection of organisms and habitats in invasion management (Pyšek et al., 2008). Woody species, such as those from genera *Tamarix* or *Prosopis*, are listed among the most damaging invasives worldwide (Lowe et al., 2000), possibly explaining why these species, and their associated habitats, received higher focus from the beginning of remote sensing applications (Jarnevich et al., 2011). Notwithstanding, a shift towards herbaceous invaders in production systems occurred from the 2000s (Appendix D). Herbaceous species are more likely to be targeted due to their socio-economic impacts in agriculture, cattle raising and forestry (Wilson et al., 2007). The increasing availability of remote sensing products with higher spatial, spectral and temporal resolutions also improved the accuracy of mapping and detecting invasions by herbaceous plants (Blumenthal et al., 2012; Dvořák et al., 2015).

5.4.2. Sources of remote sensing data for managing plant invasions

Most remote sensing studies in our dataset focus on later invasion stages, namely dominance (37% of all records) and expansion (23%; Figure 5.3a). Interest on using remote sensing in the management of early invasion stages, i.e. introduction (4%) and establishment (33%), was only observed in the last decade (Appendix D). Since detection capacity increases at higher spatial, spectral and radiometric resolutions, this trend may be related to the computational capacity and data requirements needed to achieve reliable results in the identification of invasive plants, particularly when they are confined to small populations and/or areas (He et al., 2011; Juanes, 2018). Such can also be related to the emergence of improved satellite remote sensing data since the mid-2000s (time until which airborne products dominated the field) and of high-resolution UAV products during the last years (Manfreda et al., 2018; Appendix D).

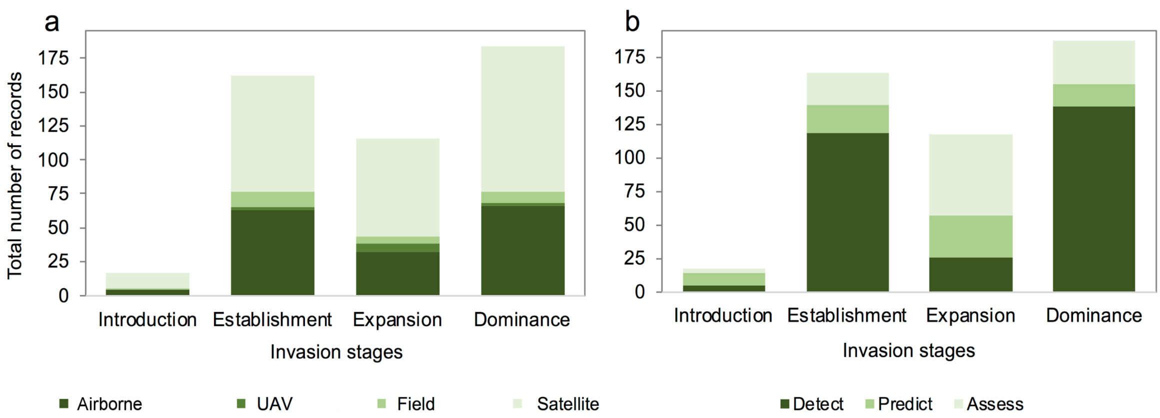

Figure 5.3. The number of records applying remote sensing to the management of plant invasions (from 1977 to 2016) distributed through the stages of the invasion process, considering: (a) remote sensing data sources, showing the ability of satellite and airborne data for managing plant invasion at late and early invasion stages; and (b) remote sensing contribution to management, highlighting the capacity for detecting plant invasions at late invasion stages, for predicting invasions at early stages, and for assessing invasion impacts. See Appendix D for the detailed representation of temporal trends in published records.

The use of satellite derived products in invasion management is still the most prevalent, regardless of the invasion stage. Satellite products make up 52% of all studies, followed by airborne (34%), field measurements (e.g. through field spectrometers; 11%) and UAV (2%). A large proportion of studies that include satellite sources rely on data derived from Landsat TM/ETM+ sensors (21% of all records), followed by a minor proportion from MODIS (7%), Hyperion-EO (5%), Quickbird, and Ikonos (4% each; Appendix D). Most satellite approaches evaluate land cover changes caused by the expansion of invasive species (using time-series analysis and multispectral imagery) or refine methodologies for improving the spectral discrimination of dominant invasive vegetation (e.g. Alvarez-Taboada et al., 2017; Hauglin and Ørka, 2016; Peerbhay et al., 2016).

Despite the use of multispectral data in invasion studies from the early 2000s (55% of all studies; possibly due to the increase of publicly available imagery), hyperspectral (35% of all records) and radar (9%) information only grew during the last decade (Appendix D). This increasing interest in more refined remote sensing data seems to mostly target later invasion stages, as well. The slow implementation of this type of data in the assessment of earlier stages of invasion and impacts, may be explained by the high cost and difficulty of acquiring and processing hyperspectral and radar data from both satellite and airborne sources.

However, current trends for providing high spatial and spectral resolution data (especially from older image libraries, such as the Hyperion-EO dataset) can contribute to their increased application, particularly in understanding invasion impacts (He et al., 2011; Thamaga and Dube, 2018).

5.4.3. Contributions from remote sensing to the management of plant invasions

The amount of remote sensing studies focused on detecting plant invasions at late invasion stages (57% of all records) largely overcomes the set of studies focused on the prediction of invasion processes at earlier stages (15%) and on the assessment of impacts during invasion expansion (28%; Figure 5.3b). Since its beginning, remote sensing has particularly contributed to detect plant invaders, from the discrimination of spectral signatures (e.g. Grosse-Stoltenberg et al., 2016; Lehmann et al., 2015) to the identification of dominant or already established invasive populations (e.g. Alvarez -Taboada et al., 2017; Hill et al., 2017; Michez et al., 2016; Peerbhay et al., 2016).

The use of remote sensing to help managers on predicting invasion occurrence and spread in space and through time has only emerged over the last 15 years (Figure 5.3b; see also Appendix D). The growth of this predictive capacity seems to be consistent with the general development of ecological modelling from that time onwards (Guisan and Zimmermann, 2000), with increasing predictions of species distributions based on both static and dynamic models (Buchadas et al., 2017). Invasion studies with predictive outputs mostly focused on the acquirement of remote sensing information as predictors or response variables (Jarnevich et al., 2011; He et al., 2011, 2015; Rocchini et al., 2015). Examples include the use of models to quantify relationships between the cover of plant invasions (spatially digitised from very high-resolution aerial imagery) and ancillary environmental and disturbance predictors (Blumenthal et al., 2012; López and Stokes, 2016); the inclusion of remotely sensed variables (e.g. geospatial information) as predictors of species distributions and impacts (e.g. Hellmann et al. 2017); or the use of remotely sensed indices (e.g. from phenocams or satellite sensors) as response variables in modelling frameworks (e.g. Morisette et al., 2006; West et al, 2016).

The capacity of remote sensing to assess invasions, and, particularly to support managers with decisions that could mitigate invasion impacts, has only become evident with the increasing access to multi-temporal satellite images (see section 5.3). These assessments have provided managers with effective tools for the evaluation of changes in invasion patterns and phenology through time (e.g. Michez et al., 2016; Richardson et al., 2018), alterations of the invasion process due to biocontrol or restoration efforts (e.g. Bedford et al., 2018; Ji et al., 2017; Nagler et al., 2014), or aspects related to the impacts of plant invasions on ecosystem functions or properties (e.g. Espinar et al., 2015; Dzikiti et al., 2016), including the anticipation of regime shifts in ecosystem functions (Große-Stoltenberg et al., 2018).

Thus, the study of invasions has expanded from detecting already established invasions, towards predicting new invasion processes and assessing invasion-induced changes. It also evolved from using only raw spectral and land cover information, towards combining multiple remote sensing products dealing with vegetation (with 68% of the records), water (12%), topography (10%), soil (7%), and climate (3%; see Appendix D). Among these products, the most applied have a biophysical meaning related to the carbon and energy dynamics (Cabello et al., 2012), such as NDVI (*Normalized Difference Vegetation Index*), EVI (*Enhanced Vegetation Index*), NDWI (*Normalized Difference Water Index),* SAVI (*Soil-Adjusted Vegetation Index*), LAI (*Leaf Area Index*), LST (*Land Surface Temperature*) and vegetation height (as radar output; see also Appendix D). Integrating biophysically meaningful remote sensing products can be particularly relevant for managing invasion impacts, by providing a link among ecological processes, functions and services (e.g. Alcaraz-Segura et al., 2013). As an example, the use of colour indices extracted from phenocam imagery has been increasingly used to monitor changes in vegetation phenology and hence ecosystem impacts (Richardson et al., 2018). Also, remotely-sensed *Ecosystem Functional Attributes* (EFAs), which describe the exchanges of matter and energy between biota and the physical environment (e.g. indicators of productivity, seasonality, and phenology; Alcaraz-Segura et al., 2009; Pettorelli et al., 2017), are particularly interesting, constituting *Essential Biodiversity Variables* (EBVs; Alcaraz-Segura et al., 2017) with critical information for managing biodiversity and overall natural resources (Pettorelli et al., 2016; Kissling et al. 2017).

In sum, our review suggests a duality of approaches in the application of remote sensing to the management of plant invasions. This duality is also illustrated by a PCA of the records retrieved from our search (Figure 5.4). On the one hand, remote sensing has been used to detect invasive species, particularly at developed invasion stages and through airborne instruments (cf. PCA group on the right, Figure 5.4). The discrimination of spectral signatures of invasive plants requires detailed resolutions (spectral, temporal, spatial and radiometric), which can be more easily obtained through airborne technology (Toth and Jóźków, 2016), and have been more often applied at later invasion stages (Juanes, 2018). On the other hand, remote sensing has been applied in the prediction of early invasion stages and in the assessment of invasion impacts, based on satellite-derived indices and products with biophysical meaning (cf. PCA group on the left, Figure 5.4). Effective management options need to prevent, as well as to anticipate the extension of potential impacts, particularly when invasive species are still expanding. Satellite remote sensing is useful in this regard (Smith et al., 2017), providing multi-temporal and large-scale information with relevant ecological meaning (e.g. EFAs and EBVs; Kissling et al. 2017; Pettorelli et al., 2016; West et al., 2016), namely through indicators currently under development in global initiatives such as GEOBON (at: http://geobon.org/). Also promising is the increasing development of indices extracted from phenocam imagery (Richardson et al., 2018). Especially when considered in a network (e.g. *US National Ecological Observatory Network*; at: https://www.neonscience.org/), this technology can offer high-frequency data on vegetation phenology, useful for model validation and satellite data calibration across wide regions of interests (Browning et al., 2017; Richardson et al., 2018; Yan et al., 2017).

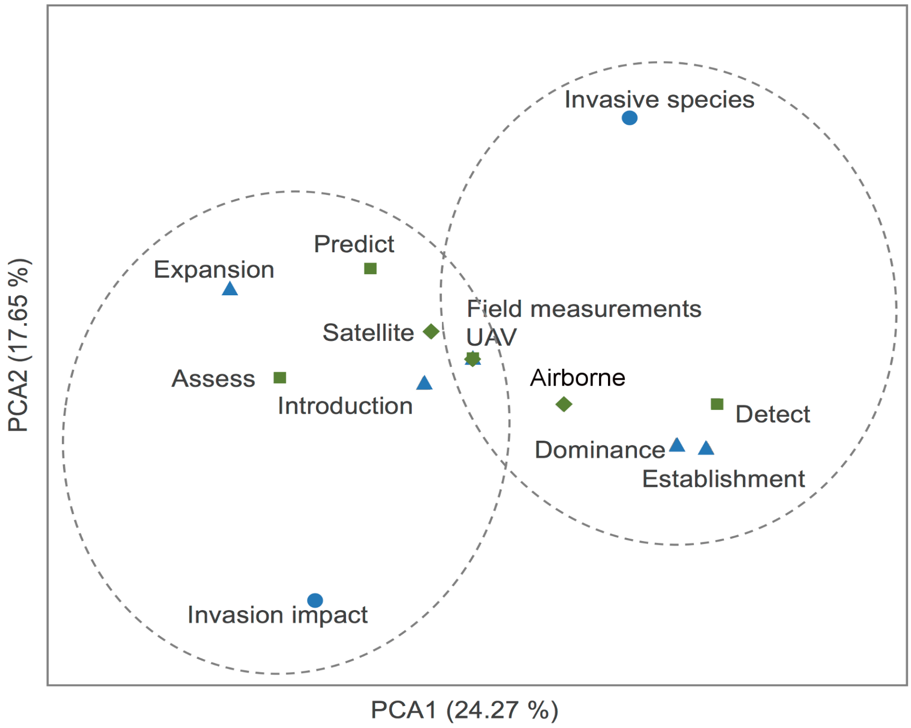

Figure 5.4. Principal Components Analysis (PCA) of the records classified according to “invasion process” (i.e. introduction, establishment, expansion, dominance - represented by triangles), “management focus” (focus on the species or on their impacts - circles), “remote sensing sources” (field measurements, airborne, UAV, or satellites - diamonds), and “remote sensing contributions” (detect, predict, assess - squares). Time-period: 1977-2016. Number of records: 289. A duality of approaches is suggested by the drawn dashed lines: prediction of early invasion stages and assessment of invasion impacts, based on satellite information (on the lower left); and detection of plant species through airborne instruments, especially when at developed invasion stages groups (on the upper right). Values in brackets refer to the amount of variance explained by PCA axes 1 and 2.

5.5. LIMITATIONS AND OPPORTUNITIES OF REMOTE SENSING IN INVASION MANAGEMENT

5.5.1. Remote sensing limitations

Since the beginning of the application of remote sensing on plant invasions, insufficient spectral resolution (29% of all records) and technical constraints (16%) have been indicated as main limitations (Figure 5.5). The inability to adequately identify and separate invasive plants from their surrounding or background environments, particularly in biodiverse ecosystems, constitutes a main challenge for an accurate detection (Bradley, 2014). Constraints highlighted in our records also include limitations in spectral resolutions in the most common (and inexpensive) airborne and satellite platforms (e.g. Andrew and Ustin, 2008), and insufficient phenological differences among species for an accurate discrimination from heterogeneous backgrounds (e.g. Hudson et al., 2015).

Limitations related to spatial resolution and data availability (and costs) become more evident since the early 2000s (Figure 5.5). Despite the broader range of available satellite products, low spatial resolution was still recognised as the main cause for classification inaccuracies when discriminating invaders. Also, the high cost of remote sensing equipment required for more appropriate resolutions (airplanes, drones, hyperspectral satellites) was often mentioned (Ramsey III et al., 2005). Conversely, the use of airborne platforms (both airborne and UAV) with higher discrimination accuracies, would potentially fail in covering all areas in which management was necessary. It would also need expertise and skills that are logistically and technically demanding, such as trained technicians, specialised services and software, and powerful processing equipment (e.g. Manfreda et al., 2018; Underwood et al., 2003).

In fact, technical constraints were increasingly indicated as limitations from the 2000s (Figure 5.5). In line with higher data availability, reported limitations also concerned the speed of data analysis and transfer, as well as the amount of space required for data storage (Blumenthal et al., 2012). Technical training and skills become even more necessary, in order to avoid inaccuracies in invasive species classification from satellite-derived data, e.g. due to shadows, view angle variability, sub-pixel cloud cover, sensor calibration, remnant geometric errors, among others (Huang and Asner, 2009; Thamaga and Dube, 2018). Even with the higher spatial resolution and accessibility to UAV platforms, the capability of batteries and time flight is generally too low to consider UAVs practical for repeated monitoring of invasive plants over large areas (Sankey et al., 2017). Surveying restrictions regarding the operation of drones in aerial platforms (such as the A-NPA 2015-10 regulatory framework in Europe) were also mentioned as potential limitations (e.g. Calviño-Cancela et al., 2014).

Our survey suggests a recent awareness of limitations in temporal coverage and field data validation (from the 2010s; Figure 5.5). Temporal gaps in available satellite imagery and strong temporal variations (phenology) of invasive species can lead to inaccurate estimations; therefore, understanding phenological differences across species would be strongly recommended (Hudson et al., 2015; Müllerová et al., 2013, 2017a). Academics also seem to realize the advantages of "*keeping your feet on the ground*", by recommending more field surveys and validation (Asner et al., 2010; He et al., 2011, 2015; Manfreda et al., 2018), as well as the integration of complementary sources of information (i.e. coupled satellite, phenocam, LiDAR and ground observations; Browning et al., 2017; Sankey et al., 2017; Yan et al., 2017) to support remote sensing analyses. Devaluing the role of ground truth (both training and test data; Evangelista et al., 2018) and disregarding the publication of field-collected information has made many academics and managers uncertain when assessing invaders or evaluating their impacts on invaded habitats (e.g. Albright, 2004), and when estimating the efficiency of management actions (Nagler et al., 2014).

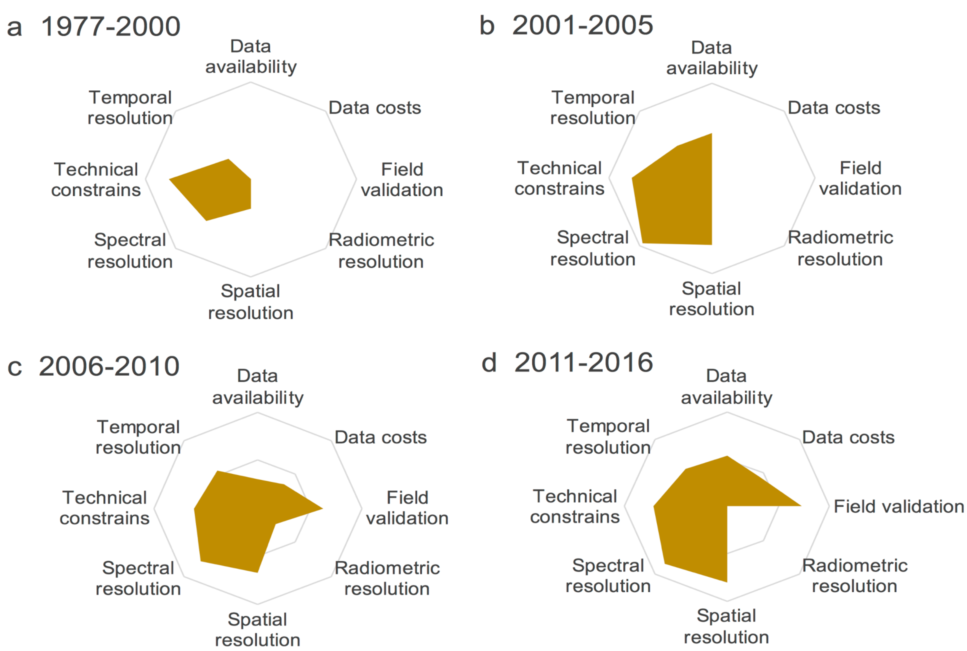

Figure 5.5. Radar plots illustrating the proportion of records (logarithmic scale) from the search considering different remote sensing limitations, including: (a) spectral resolution and technical constraints since the 1970s; (b) growing concerns with spatial resolution and data availability (and costs) in the early 2000s; (c) increasing technical, temporal, and field validation restrictions (late 2000s); and (d) awareness of the wide diversity of limitations when dealing with plant invasions (from the 2010s).

5.5.2. Advancing invasion management through remote sensing

Despite current limitations, remote sensing offers a series of opportunities to target challenges underlying the management of plant invasions (Alcaraz-Segura et al., 2013, 2017; Cabello et al., 2012; Pettorelli et al., 2014; Rocchini et al., 2015; Figure 5.6). Managing plant invasions from space needs to move from merely identifying "*the invasion by a given species in a particular ecosystem*", to predicting and assessing "*interlinked invasion impacts in a given region*" (after Vaz et al., 2017b), focusing on the most relevant traits, populations or communities to manage from a functional perspective (especially when not all species can be remotely detected). This would still allow focusing on invasive species and invaded habitats, while integrating useful insights from interdisciplinarity perspectives, e.g. impacts on ecosystem services (Figure 5.6; Andrew et al., 2014; Niphadkar and Nagendra, 2016). In fact, applications of remote sensing through the lens of ecosystem services could constitute an opportunity to manage invasions in an integrated and efficient manner, focusing on understanding and managing ecosystem dynamics, in contrast to targeting only individual species (Abelleira Martínez et al., 2016; Andrew et al., 2014).

Remote sensing data can be particularly useful when applied in modelling frameworks, by providing both explanatory and response variables, and thus assisting on the anticipation, early-detection and prediction of invasive species, invaded areas and respective impacts (Große-Stoltenberg et al., 2018; He et al., 2015; Juanes, 2018; Rocchini et al., 2015). It can improve prevention and/or eradication actions at early invasion stages (Juanes, 2018; Simberloff et al., 2013), thus contributing to protect ecosystem services and/or avoid potential ecosystem disservices that could result from invasions (Vaz et al., 2017a). When plant invaders are already established or expanding, remote sensing can be used to early-detect potential impacts and evaluate the extent of invasions, supporting management measures aiming at mitigating and treating the impacts of these species on ecosystem functioning and services (Bradley, 2014; Dzikiti et al., 2016; Große-Stoltenberg et al., 2018; Pettorelli et al., 2017). Also, the potential of remote sensing to evaluate changes in essential ecosystem functional variables linked to species distributions (Alcaraz-Segura et al., 2017; Pettorelli et al., 2014, 2017; West et al., 2016) makes it a valuable tool for assessing biophysical changes driven by invaders. Overall, these contributions from remote sensing are of utmost importance not only to monitor invasions, but also to mitigate, restore and adapt to their potential impacts (Simberloff et al., 2013; Vaz et al., 2017a; Figure 5.6).

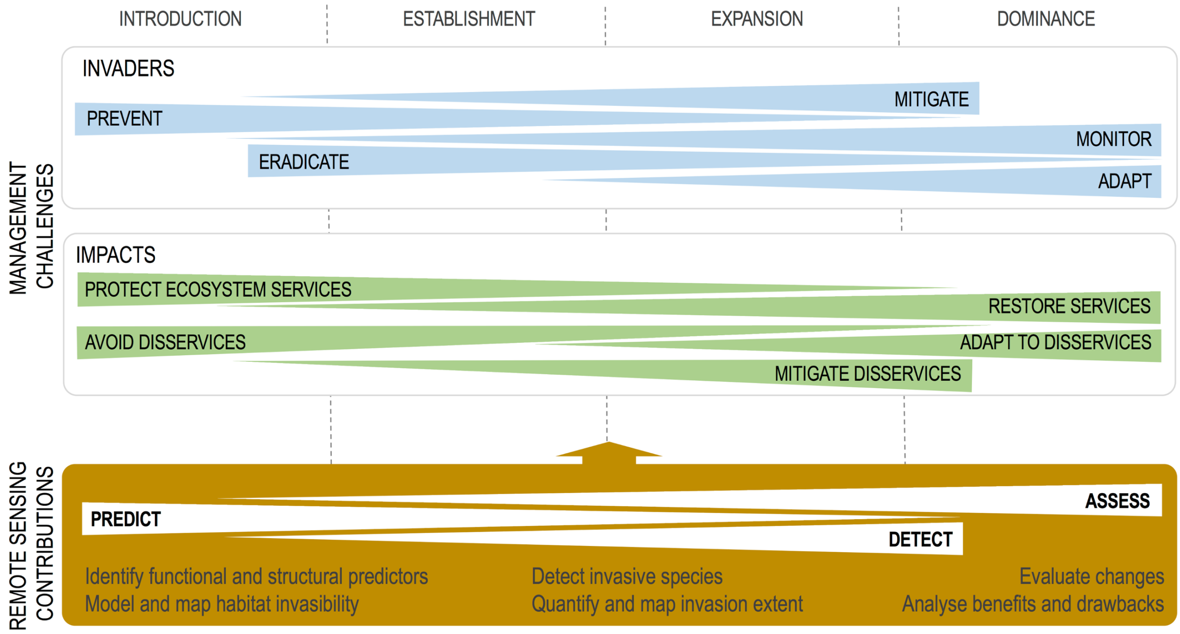

Figure 5.6. A framework for advancing invasion research through the lens of remote sensing. The framework considers the challenges of managing invaders and their impacts, based on the suggested opportunities from remote sensing: (1) prediction: remote sensing linked to modelling frameworks to predict invasive species and invaded areas, hence to support prevention and eradication actions at early invasion stages to ensure protection of ecosystem services and avoid ecosystem disservices; (2) detection: remote sensing applied to detect and evaluate the extent of invasions in introduced areas, as well as to guide management options that mitigate and treat invasions and their impacts; and (3) assessment: remote sensing as a tool for assessing changes in invasion dominance and impacts, allowing at monitoring invasions and mitigating, restoring and adapting to changes. The heights of drawn triangles represent different effort levels across invasion stages.

In order to make remote sensing more operational, and to further support management actions focused on the impacts of plant invasions on ecosystem services (and disservices), we highlight the need of: (a) motivating field work and experimental studies that can support accuracy in remote sensing (e.g. Pau and Dee, 2016; Evangelista et al., 2018); (b) acquiring deeper knowledge on species phenology and ecosystem dynamics (e.g. Murray et al., 2018; Richardson et al., 2018); (c) applying for free access and using already available multi-temporal imagery and tools (e.g. Kwok, 2018; Visser et al., 2014), considering multiple scenes and seasonal dynamics, together with ancillary data in already developed modelling platforms (Browning et al., 2017; Thamaga and Dube, 2018); (d) pursuing statistical and computational solutions (e.g. Rocchini et al., 2015; Skidmore and Pettorelli, 2015) that allow the transferability of results across spatial and temporal scales, such as automated processing and classification algorithms focused on available imagery at distinct time periods (He et al., 2015); and (e) making scientific information available (Turner et al., 2015), including field validation data, spectral libraries, and remote sensing outputs.

In this context, future development of remote sensing will open new opportunities for managers and decision-makers to make invasion management more operational, by improving our capacity to detect, predict, and assess invasions and their impacts on ecosystems and their services. Improvements will comprise higher availability of multispectral optical imagery with increasing spatial and temporal resolutions, including the use of new generation sensors (e.g. Landsat-8, Sentinel-2), upcoming satellites such as Digital Globe, Planet Labs, Black Sky or Airbus Defence and Space, UAV platforms with multispectral, hyperspectral and thermal imaging (Juanes, 2018; Manfreda et al., 2018; Müllerová et al., 2017b; Thamaga and Dube, 2018), and the expansion or creation of satellite and ground-based observation networks for monitoring vegetation phenology at unprecedented frequencies (Browning et al., 2017; Richardson et al., 2018). Alongside data availability, there is a rising effort on developing open-source and user-friendly platforms with increasing processing power and speed (e.g. Google Earth Engine or the online tools Remap and AppEEARS; Kwoks, 2018).

The integration of information from different remote sensing sensors (mounted either on-board satellite, airborne or ground platforms; Browning et al., 2017; Nagler et al., 2014; Yan et al., 2017) and their combination with data sources from other disciplines (e.g. social media, citizen science, molecular information; e.g. Kissling et al., 2017; Toth and Józków, 2016) and with novel computer processing approaches and algorithms (He et al., 2015), such as data-fusion techniques (Joshi et al., 2016) and artificial intelligence (Guirado et al., 2017), will have a high potential for improving our learning about plant invaders and their impacts. For instance, the upcoming German EnMAP hyperspectral mission will offer unprecedented spatial (up to 30 m), temporal (minimum revisit times of 21 days) and spectral (220 spectral bands) resolution images (Palubinskas et al., 2017). The expected availability and processing of Lidar 3D data (both from airborne multispectral sensors and from the GEDI mission on-board the *International Space Station*) will allow to detect and predict invasive plants, for instance, under forest canopies (Hopkinson et al., 2016). This is also applicable using active and passive radar data, as well as solar-induced fluorescence data (Smith et al., 2017), which constitute new possibilities for the assessment of plant invasions and their impacts, e.g. based on soil moisture (e.g. SMAP mission; Mohanty et al., 2017) or photosynthesis dynamics (e.g. from the FLEX mission; Sun et al., 2017).

5.6. CONCLUSIONS

We have reviewed and discussed the progress, current state and opportunities of remote sensing in the study and management of plant invasions. Our quantitative review revealed that the application of remote sensing to the management of plant invasions has become prominent during the last couple of decades, mostly as a result of advances in technology and the evolution of invasion science. Overall, remote sensing has been contributing to the detection of invasive species, prediction of invasion patterns and dynamics, and assessment of invasion impacts. For the detection of invasive species, the use of airborne instruments has been particularly useful at advanced invasion stages (i.e. expansion and dominance). This emphasises that the discrimination of spectral signatures of invasive plants often requires highly detailed resolutions (spectral, temporal and spatial), which can be obtained through airborne technology. Also, remote sensing has been applied both to predict early invasion stages and to assess invasion impacts, mainly based on satellite information. In this regard, satellite-derived variables have been useful to predict and evaluate invasion impacts, being particularly helpful in the anticipation and prevention of new invasions and further impacts, mainly when species are still establishing.

Despite its limitations, remote sensing is progressing, as technology evolves, to develop interdisciplinary strategies for a more efficient and better use of available options in the management of plant invasions. Considering current and future Earth observation missions, remote sensing includes a set of opportunities to improve the management of plant invasions, namely to: (a) feed modelling frameworks for the prediction of invasive species distributions and dynamics, and hence support prevention and/or eradication of early invasion stages, while protecting ecosystem services (and avoiding disservices); (b) detect the extent of invasions when plant invaders are establishing and expanding in the new range, assisting on any management strategy aiming at mitigating and treating invasion impacts; and (c) assess changes in invasion dominance, providing important information to monitor invasions, and to mitigate, restore and adapt to their impacts. In order to make remote sensing more operational for invasion management, taking advantage of future remote sensing missions, dedicated field work and experimental studies should be encouraged, together with the current development of analytical tools and computational solutions that can be most suitable for management goals.

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SUPPLEMENTARY MATERIAL IV

Appendix A - Details on keyword selection and literature search procedure

Searches were performed considering a set of selected keywords. Following the Population-Intervention-Comparison-Outcome (PICO) strategy (Higgins and Green, 2011), search terms related to “plant invasions” (i.e., Population) and “remote sensing” (i.e., Intervention) were compiled based on a list derived from a number of core references (see below) and the expert knowledge of the research team. The research team included members with expertise in remote sensing, plant invasions, and invasion management.

The selection of the final set of keywords was done in ‘‘ISI Web of Science’’ (ISI; http:// webofknowledge.com/; in September 2016). The final search string (highlighted with bold letters, in Tables S5.1 and S5.2) was derived through an iterative procedure, by (1) reviewing a short list of key publications, and including pertinent keywords for the search, (2) checking the records retrieved by the search, and including or excluding pre-existent and new keywords, and (3) re-conducting the search with the new set of keywords. New terms were step-by-step added and then the first 10 hits were checked for relevance. If the new search results were an improvement over the old ones the new term was kept, else it was removed.

Table S5.1. Search terms related to “plant invasions” and respective number of retrieved records.

Search terms	**Number records**
"plant invader*" OR "introduced plant*" OR "Non-native plant*" OR "Nonnative plant*" OR "invasive plant*" OR "exotic plant*" OR "alien plant*" OR "plant invasion*" OR "Nonindigenous plant*" OR "non-indigenous plant*" OR "allochthonous plant*"	14690
"plant invader*" OR "introduced plant*" OR "Non-native plant*" OR "Nonnative plant*" "invasive plant*" OR "exotic plant*" OR "alien plant*" OR "plant invasion*" OR "Nonindigenous plant*" OR "non-indigenous plant*" OR "tree invader*" OR "introduced tree*" OR "Non-native tree*" OR "Nonnative tree*" "invasive tree*" OR "exotic tree*" OR "alien tree*" OR "tree invasion*" OR "Nonindigenous tree*" OR "non-indigenous tree*" OR "allochthonous tree*" OR "non-indigenous tree*" OR "allochthonous tree*"	16493
"plant invader*" OR "introduced plant*" OR "Non-native plant*" OR "Nonnative plant*" "invasive plant*" OR "exotic plant*" OR "alien plant*" OR "plant invasion*" OR "Nonindigenous plant*" OR "non-indigenous plant*" OR "tree invader*" OR "introduced tree*" OR "Non-native tree*" OR "Nonnative tree*" "invasive tree*" OR "exotic tree*" OR "alien tree*" OR "tree invasion*" OR "Nonindigenous tree*" OR "non-indigenous tree*" OR "allochthonous tree*" OR "forest invader*" OR "introduced forest*" OR "Non-native forest*" OR "Nonnative forest*" "invasive forest*" OR "exotic forest*" OR	16997

"alien forest*" OR "forest invasion*" OR "Nonindigenous forest*" OR "non-indigenous forest*" OR "allochthonous forest*"	
"plant invader*" OR "introduced plant*" OR "non-native plant*" OR "nonnative plant*" "invasive plant*" OR "exotic plant*" OR "alien plant*" OR "plant invasion*" OR "nonindigenous plant*" OR "non-indigenous plant*" OR "allochthonous plant*" OR "tree invader*" OR "introduced tree*" OR "Non-native tree*" OR "nonnative tree*" "invasive tree*" OR "exotic tree*" OR "alien tree*" OR "tree invasion*" OR "nonindigenous tree*" OR "non-indigenous tree*" OR "allochthonous tree*" OR "forest invader*" OR "introduced forest*" OR "Non-native forest*" OR "Nonnative forest*" "invasive forest*" OR "exotic forest*" OR "alien forest*" OR "forest invasion*" OR "Nonindigenous forest*" OR "non-indigenous forest*" OR "allochthonous forest*" OR "introduced vegetation*" OR "non-native vegetation*" OR "nonnative vegetation*" "invasive vegetation*" OR "exotic vegetation*" OR "alien vegetation*" OR "nonindigenous vegetation*" OR "non-indigenous vegetation*" OR "allochthonous vegetation*"	17331
"plant invader*" OR "introduced plant*" OR "non-native plant*" OR "nonnative plant*" "invasive plant*" OR "exotic plant*" OR "alien plant*" OR "plant invasion*" OR "nonindigenous plant*" OR "non-indigenous plant*" OR "allochthonous plant*" OR "tree invader*" OR "introduced tree*" OR "Non-native tree*" OR "nonnative tree*" "invasive tree*" OR "exotic tree*" OR "alien tree*" OR "tree invasion*" OR "nonindigenous tree*" OR "non-indigenous tree*" OR "allochthonous tree*" OR "forest invader*" OR "introduced forest*" OR "Non-native forest*" OR "Nonnative forest*" "invasive forest*" OR "exotic forest*" OR "alien forest*" OR "forest invasion*" OR "Nonindigenous forest*" OR "non-indigenous forest*" OR "allochthonous forest*" OR "introduced vegetation*" OR "non-native vegetation*" OR "nonnative vegetation*" "invasive vegetation*" OR "exotic vegetation*" OR "alien vegetation*" OR "nonindigenous vegetation*" OR "non-indigenous vegetation*" OR "allochthonous vegetation*" OR "shrub invader*" OR "introduced shrub*" OR "Non-native shrub*" OR "nonnative shrub*" "invasive shrub*" OR "exotic shrub*" OR "alien shrub*" OR "shrub invasion*" OR "nonindigenous shrub*" OR "non-indigenous shrub*" OR "allochthonous shrub*"	17840
"plant invader*" OR "introduced plant*" OR "non-native plant*" OR "nonnative plant*" "invasive plant*" OR "exotic plant*" OR "alien plant*" OR "plant invasion*" OR "nonindigenous plant*" OR "non-indigenous plant*" OR "allochthonous plant*" OR "tree invader*" OR "introduced tree*" OR "Non-native tree*" OR "nonnative tree*" OR "exotic tree*" OR "alien tree*" OR "nonindigenous tree*" OR "non-indigenous tree*" OR "allochthonous tree*" OR "introduced forest*" OR "non-native forest*" OR "Nonnative forest*" OR "exotic forest*" OR "alien forest*" OR "Nonindigenous forest*" OR "non-indigenous forest*" OR "allochthonous forest*" OR "introduced vegetation*" OR "non-native vegetation*" OR "nonnative vegetation*" "invasive vegetation*" OR "exotic vegetation*" OR "alien vegetation*" OR "nonindigenous vegetation*" OR "non-indigenous vegetation*" OR "allochthonous vegetation*" OR "shrub invader*" OR "introduced shrub*" OR "non-native shrub*" OR "nonnative shrub*" "invasive shrub*" OR "exotic shrub*" OR "alien shrub*" OR "shrub invasion*" OR "nonindigenous shrub*" OR "non-indigenous shrub*" OR "allochthonous shrub*" OR "herb invader*" OR "introduced herb*" OR "Non-native herb*" OR "nonnative herb*" "invasive herb*" OR "exotic herb*" OR "alien herb*" OR "herb invasion*" OR "nonindigenous herb*" OR "non-indigenous herb*" OR "allochthonous herb*"	18393
"plant invader*" OR "introduced plant*" OR "non-native plant*" OR "nonnative plant*" "invasive plant*" OR "exotic plant*" OR "alien plant*" OR "plant invasion*" OR "nonindigenous plant*" OR "non-	18422

indigenous plant*" OR "allochthonous plant*" OR "tree invader*" OR "introduced tree*" OR "Non-native tree*" OR "nonnative tree*" "invasive tree*" OR "exotic tree*" OR "alien tree*" OR "tree invasion*" OR "nonindigenous tree*" OR "non-indigenous tree*" OR "allochthonous tree*" OR "forest invader*" OR "introduced forest*" OR "non-native forest*" OR "Nonnative forest*" "invasive forest*" OR "exotic forest*" OR "alien forest*" OR "forest invasion*" OR "Nonindigenous forest*" OR "non-indigenous forest*" OR "allochthonous forest*" OR "introduced vegetation*" OR "non-native vegetation*" OR "nonnative vegetation*" "invasive vegetation*" OR "exotic vegetation*" OR "alien vegetation*" OR "nonindigenous vegetation*" OR "non-indigenous vegetation*" OR "allochthonous vegetation*" OR "shrub invader*" OR "introduced shrub*" OR "non-native shrub*" OR "nonnative shrub*" "invasive shrub*" OR "exotic shrub*" OR "alien shrub*" OR "shrub invasion*" OR "nonindigenous shrub*" OR "non-indigenous shrub*" OR "allochthonous shrub*" OR "herb invader*" OR "introduced herb*" OR "Non-native herb*" OR "nonnative herb*" "invasive herb*" OR "exotic herb*" OR "alien herb*" OR "herb invasion*" OR "nonindigenous herb*" OR "non-indigenous herb*" OR "allochthonous herb*" OR "introduced landscape" OR "non-native landscape" OR "nonnative landscape" OR "invasive landscape" OR "exotic landscape" OR "alien landscape" OR "nonindigenous landscape" OR "non-indigenous landscape" OR "allochthonous landscape"	
"plant invader*" OR "introduced plant*" OR "non-native plant*" OR "nonnative plant*" OR "invasive plant*" OR "exotic plant*" OR "alien plant*" OR "plant invasion*" OR "nonindigenous plant*" OR "non-indigenous plant*" OR "allochthonous plant*" OR "tree invader*" OR "introduced tree*" OR "Non-native tree*" OR "nonnative tree*" OR "invasive tree*" OR "exotic tree*" OR "alien tree*" OR "tree invasion*" OR "nonindigenous tree*" OR "non-indigenous tree*" OR "allochthonous tree*" OR "forest invader*" OR "introduced forest*" OR "non-native forest*" OR "Nonnative forest*" "invasive forest*" OR "exotic forest*" OR "alien forest*" OR "forest invasion*" OR "Nonindigenous forest*" OR "non-indigenous forest*" OR "allochthonous forest*" OR "introduced vegetation*" OR "non-native vegetation*" OR "nonnative vegetation*" OR "invasive vegetation*" OR "exotic vegetation*" OR "alien vegetation*" OR "nonindigenous vegetation*" OR "non-indigenous vegetation*" OR "allochthonous vegetation*" OR "shrub invader*" OR "introduced shrub*" OR "non-native shrub*" OR "nonnative shrub*" OR "invasive shrub*" OR "exotic shrub*" OR "alien shrub*" OR "shrub invasion*" OR "nonindigenous shrub*" OR "non-indigenous shrub*" OR "allochthonous shrub*" OR "herb invader*" OR "introduced herb*" OR "Non-native herb*" OR "nonnative herb*" OR "invasive herb*" OR "exotic herb*" OR "alien herb*" OR "herb invasion*" OR "nonindigenous herb*" OR "non-indigenous herb*" OR "allochthonous herb*" OR "introduced landscape" OR "non-native landscape" OR "nonnative landscape" OR "invasive landscape" OR "exotic landscape" OR "alien landscape" OR "nonindigenous landscape" OR "non-indigenous landscape" OR "allochthonous landscape" OR "novel ecosystem*"	**18886**

Table S5.2. Search terms related to “remote sensing” and respective number of retrieved records.

Search terms	**Number records**
"Remote* sens*" OR "earth observation"	122
"Remote* sens*" OR "earth observation"OR imagery OR *radiometer OR radiometry	167
"Remote* sens*" OR "earth observation"OR imagery OR *radiometer OR radiometry OR satellite* OR UAV OR drone OR"unmanned aerial" OR aircraft* OR AVHRR OR sensor* OR radar OR Modis OR lidar	204
"Remote* sens*" OR "earth observation"OR imagery OR *radiometer OR radiometry OR satellite* OR UAV OR drone OR"unmanned aerial" OR aircraft* OR AVHRR OR sensor* OR radar OR Modis OR lidar OR sentinel* OR landsat*	229
"Remote* sens*" OR "earth observation"OR imagery OR *radiometer OR radiometry OR satellite* OR UAV OR drone OR"unmanned aerial" OR aircraft* OR AVHRR OR sensor* OR radar OR Modis OR lidar OR sentinel* OR landsat* OR "high spatial resolution" OR *spectral OR "image* fusion" OR "spectral ind*" OR NDVI	252
"Remote* sens*" OR "earth observation"OR imagery OR *radiometer OR radiometry OR satellite* OR UAV OR drone OR"unmanned aerial" OR aircraft* OR AVHRR OR sensor* OR radar OR Modis OR lidar OR sentinel* OR landsat* OR "high spatial resolution" OR *spectral OR "image* fusion" OR "spectral ind*" OR NDVI OR "enhanced vegetation index" OR ikonos* OR geoeye* OR worldview* OR pleiades* OR skysat* OR quickbird* OR triplesat* OR terrasar* OR kompsat* OR "advanced spaceborne thermal emission and reflection radiometer" OR "satellite pour l’observation de la terre"	253
"remote* sens*" OR "remote-sens*" OR "earth observation" OR "imagery" OR "UAV" OR "drone" OR "unmanned aerial" OR "aircraft*" OR "airborne" OR "air-borne" OR "spaceborne" OR "space-borne" OR "AVHRR" OR "radiomet*" OR "high-resolution" OR "high resolution" OR "very-high resolution" OR "high spatial resolution" OR "very-high spatial resolution" OR "hyper-spectral" OR "hyperspectral" OR "multispectral" OR "multi-spectral" OR "image* fusion" OR "NDVI" OR "satellite*" OR "sensor*" OR radar OR "MODIS" OR LiDAR OR "sentinel*" OR "landsat*" OR "worldview*"	**268**

The final set of keywords relating to “plant invasions” and “remote sensing” were combined with the boolean string “AND” and were used to perform the literature search in the field ‘topic’ or “tilte+abstract+keywords” of ISI Web of Science and Scopus. We considered records written in English, French, Spanish or Portuguese.

We also conducted a broader search in 92 different publication sources specialised in remote sensing, in order to detect situations in which the name of the species would be mentioned in the title, abstract or keywords, without referring to any of the terms presented in Tables S5.1. The list of additional sources considered was:

- Canadian Journal of Remote Sensing
- European Journal of Remote Sensing

- Giscience & Remote Sensing
- Ieee Geoscience and Remote Sensing Letters
- Ieee Journal of Selected Topics in Applied Earth Observations and Remote Sensing
- Ieee Transactions on Geoscience and Remote Sensing
- International Journal of Remote Sensing
- Isprs Journal of Photogrammetry and Remote Sensing
- Journal of Applied Remote Sensing
- Journal of the Indian Society of Remote Sensing
- Photogrammetric Engineering and Remote Sensing
- Remote Sensing
- Remote Sensing Letters
- Remote Sensing of Environment
- International Journal of Applied Earth Observation and Geoinformation
- International Journal of Remote Sensing
- International Geoscience and Remote Sensing Symposium (IGARSS)
- Remote Sensing Reviews
- International Archives of the Photogrammetry, Remote Sensing and Spatial Information Sciences - ISPRS Archives
- Advances in Land Remote Sensing: System, Modeling, Inversion and Application
- 2011 Joint Urban Remote Sensing Event, JURSE 2011 - Proceedings
- Earth Science Satellite Remote Sensing: Science and Instruments
- Advances in Photogrammetry, Remote Sensing and Spatial Information Sciences: 2008 ISPRS Congress Book
- Egyptian Journal of Remote Sensing and Space Science
- 2011 Microwaves, Radar and Remote Sensing Symposium, MRRS-2011 - Proceedings
- Proceedings of the Third International Workshop on the Analysis of Multi-Temporal Remote Sensing Images 2005
- 2008 Proceedings of Microwaves, Radar and Remote Sensing Symposium, MRRS 2008
- Earth Science Satellite Remote Sensing: Data, Computational Processing, and Tools
- 2006 IEEE MicroRad Proceedings - 9th Specialist Meeting on Microwave Radiometry and Remote Sensing Applications, MicroRad'06
- 2011 6th International Workshop on the Analysis of Multi-Temporal Remote Sensing Images, Multi-Temp 2011 - Proceedings
- 11th Specialist Meeting on Microwave Radiometry and Remote Sensing of the Environment, MicroRad 2010 - Proceedings
- Earth Observation and Remote Sensing
- IEEE Geoscience and Remote Sensing Magazine
- International Workshop on Microwaves, Radar and Remote Sensing, MRRS 2005
- Proceedings, 33rd International Symposium on Remote Sensing of Environment, ISRSE 2009
- Proceedings of International Conference on Computer Vision in Remote Sensing, CVRS 2012
- Proceedings of the 2012 Tyrrhenian Workshop on Advances in Radar and Remote Sensing: From Earth Observation to Homeland Security, TyWRRS 2012

- 2011 International Conference on Remote Sensing, Environment and Transportation Engineering, RSETE 2011 - Proceedings
- 2009 Joint Urban Remote Sensing Event
- 2008 International Workshop on Education Technology and Training and 2008 International Workshop on Geoscience and Remote Sensing, ETT and GRS 2008
- 2010 2nd IITA International Conference on Geoscience and Remote Sensing, IITA-GRS 2010
- 2005 IEEE Workshop on Remote Sensing of Atmospheric Aerosols
- Proceedings of the 26th Canadian Symposium on Remote Sensing
- 2007 Urban Remote Sensing Joint Event, URS
- Proceedings of MultiTemp 2007 - 2007 International Workshop on the Analysis of Multi-Temporal Remote Sensing Images
- 2008 International Workshop on Earth Observation and Remote Sensing Applications, EORSA
- Proceedings of the 2008 2nd Workshop on USE of Remote Sensing Techniques for Monitoring Volcanoes and Seismogenic Areas, USEReST 2008
- WHISPERS '09 - 1st Workshop on Hyperspectral Image and Signal Processing: Evolution in Remote Sensing
- 2nd Workshop on Hyperspectral Image and Signal Processing: Evolution in Remote Sensing, WHISPERS 2010 - Workshop Program
- 2011 International Workshop on Multi-Platform/Multi-Sensor Remote Sensing and Mapping, M2RSM 2011
- International Conference on Electric Power Systems, High voltages, Electric machines, International conference on Remote sensing - Proceedings
- Workshop on Hyperspectral Image and Signal Processing, Evolution in Remote Sensing United States
- 32nd Asian Conference on Remote Sensing 2011, ACRS 2011
- Proceedings of the 2nd International Workshop on Earth Observation and Remote Sensing Applications, EORSA 2012
- Joint Urban Remote Sensing Event 2013, JURSE 2013
- 13th Specialist Meeting on Microwave Radiometry and Remote Sensing of the Environment, MicroRad 2014 - Proceedings
- American Society for Photogrammetry and Remote Sensing - 20th Biennial Workshop on Aerial Photography, Videography, and High Resolution Digital Imagery for Resource Assessment 2005
- American Society for Photogrammetry and Remote Sensing Annual Conference 2009, ASPRS 2009
- American Society for Photogrammetry and Remote Sensing Annual Conference 2010: Opportunities for Emerging Geospatial Technologies
- American Society for Photogrammetry and Remote Sensing - ASPRS Annual Conference 2007: Identifying Geospatial Solutions
- American Society for Photogrammetry and Remote Sensing Annual Conference 2011
- American Society for Photogrammetry and Remote Sensing - ASPRS Annual Conference 2008 - Bridging the Horizons: New Frontiers in Geospatial Collaboration
- American Society for Photogrammetry and Remote Sensing - Annual Conference 2005 - Geospatial Goes Global: From Your Neighborhood to the Whole Planet
- American Society for Photogrammetry and Remote Sensing - Annual Conference of the American Society for Photogrammetry and Remote Sensing 2006: Prospecting for Geospatial Information Integration

- 30th Asian Conference on Remote Sensing 2009, ACRS 2009
- 2012 IAPR Workshop on Pattern Recognition in Remote Sensing, PRRS 2012
- American Society for Photogrammetry and Remote Sensing Annual Conference 2012, ASPRS 2012
- 28th Asian Conference on Remote Sensing 2007, ACRS 2007
- 34th Asian Conference on Remote Sensing 2013, ACRS 2013
- International Conference on Remote Sensing, Environment and Transportation Engineering, RSETE 2013
- 3rd International Workshop on Earth Observation and Remote Sensing Applications, EORSA 2014 - Proceedings
- 2014 IEEE Microwaves, Radar and Remote Sensing Symposium, MRRS 2014 - Proceedings
- Asian Association on Remote Sensing - 26th Asian Conference on Remote Sensing and 2nd Asian Space Conference, ACRS 2005
- 2012 2nd International Conference on Remote Sensing, Environment and Transportation Engineering, RSETE 2012 - Proceedings
- Proceeding - ICARES 2014: 2014 IEEE International Conference on Aerospace Electronics and Remote Sensing Technology
- 34th International Symposium on Remote Sensing of Environment - The GEOSS Era: Towards Operational Environmental Monitoring
- Asian Association on Remote Sensing - 27th Asian Conference on Remote Sensing, ACRS 2006
- Proceedings, 32nd International Symposium on Remote Sensing of Environment: Sustainable Development Through Global Earth Observations
- 33rd Asian Conference on Remote Sensing 2012, ACRS 2012
- 31st Asian Conference on Remote Sensing 2010, ACRS 2010
- 29th Asian Conference on Remote Sensing 2008, ACRS 2008
- 2012 12th Specialist Meeting on Microwave Radiometry and Remote Sensing of the Environment, MicroRad 2012 - Proceedings
- 2010 IAPR Workshop on Pattern Recognition in Remote Sensing, PRRS 2010 United States
- Proceedings of the 5th WSEAS International Conference on Remote Sensing, REMOTE '09 Greece
- 2008 IAPR Workshop on Pattern Recognition in Remote Sensing, PRRS 2008
- Proceedings, 31st International Symposium on Remote Sensing of Environment, ISRSE 2005: Global Monitoring for Sustainability and Security
- 2008 Microwave Radiometry and Remote Sensing of the Environment - 10th Specialist Meeting, Proceedings, MICRORAD
- American Society for Photogrammetry and Remote Sensing Annual Conference, ASPRS 2013
- MultiTemp 2013 - 7th International Workshop on the Analysis of Multi-Temporal Remote Sensing Images: "Our Dynamic Environment", Proceedings
- American Society for Photogrammetry and Remote Sensing - 28th Canadian Symposium on Remote Sensing and ASPRS Fall Specialty Conference 2007 United States
- EAGE/GRSG Remote Sensing Workshop
- 35th Asian Conference on Remote Sensing 2014, ACRS 2014: Sensing for Reintegration of Societies

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Appendix B - Inclusion and exclusion criteria

Inclusion and exclusion criteria applied to the main dataset, to eliminate non-relevant information for the research goals. The inclusion/exclusion criteria were applied individually to each record. Criteria were established considering the type of publication of each record: we included research articles, book chapters, book reviews, editorial material, letters, meeting abstracts, news items, notes, paper proceedings, reviews, or forum papers. We excluded biographical items, corrections/corrigendum, or records expressing messages from subjective or poetic narratives. Anonymous records were also excluded.

We filtered our records based on the PICO (Population-Intervention-Comparator-Outcome) components. Our target Population included records that focused on all non-native invasive plant species. *Non-native* plants are here defined as those plants introduced (accidentally or intentionally) by humans to new geographic areas, and *invasive plants* as non-native plants that spread, becoming abundant and leading to major impacts on the environment or society (Richardson et al., 2011). We excluded records that differed from this concept, such as clinical terms which use alien/exotic species for referring to an organism outside the human body (mostly in dentistry, ophthalmology, dermatology, oncology) or animals in laboratory experiences (clinical laboratory).

The Intervention component focused on remote sensing, defined as the acquisition of information about an object or phenomenon from a distance, i.e. without making physical contact with the object or phenomenon under observation (Campbell and Wynne, 2011). We excluded records that didn't conduct remote sensing approaches or products (e.g. cases in which the record only mentioned remote sensing, without actually focusing on it static - Comparator).

The Outcome component expressed the management of plant invasions. We excluded records that did not focus on the management of biological invasions, that focused on the outcomes of management actions for native species conservation or focused on understanding invasion dynamics when not with the clear purpose of management. We included records that apply remotes sensing for managing invasions, including control, eradication, containment, mitigation or restoration, as well as monitoring, prevention and risk assessments (Pyšek and Richardson, 2010). These criteria related to both the type of record, and the population being targeted by the record retrieved by the search.

All criteria were applied by checking the title, abstract and keywords of each record. We considered records written in English, French, Spanish or Portuguese (where the English keywords were at least referred in the title of the record).

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Appendix C - Additional information reviewed from the dataset

Besides information on the stages of invasion process, management focus, and remote sensing contributions and constrains (see Table 5.1 from the main paper), our dataset was reviewed in order to make a taxonomic and habitat characterisation, as well as to obtain more detailed information on the characteristics of remote sensing data and products being used in the reviewed papers.

Taxonomic and habitat characterisation

We identified the targeted invasive plant species and respective growth form (as herbaceous, shrubs, trees, succulents or ferns). The main habitat type under invasion was also identified based on the habitat classification scheme from IUCN (available at: http://www.iucnredlist.org/technical-documents/classification-schemes/habitats-classification-scheme-ver3).

Characteristics of remote sensing data

We identified the main source of remote sensing data being approached in the reviewed papers, as either: airborne vehicle, unmanned airborne vehicle (UAV), groundthruth (field) assessments, or satellite. The main data type was also identified, as being hyperspectral, multispectral, radar, or other type of aerial photos (i.e., without any specific characteristic).

Types of remote sensing sources

We gathered information on the main source of remote sensing information being used in the reviewed papers, as stated by the authors. Specific sources were found to be within the following list:

Aerial Photos
Aerial Videos
AISA
ALIEO-1
ALS
APHS
Aster
AToMS
AVIRIS
CASI
CBRES-Ziuan-1
CBRES-Ziuan-2
CBRES-Ziuan-3
COSMO-SkyMed-SAR
ENVISAT-ASAR
Field Spectrometer
Formosat-2
GeoEye1
GEOSAT
HJ-A
Huanjing-1
HyMap
Hyperion (not specified)
HyperionEO-1
HyperionEO-2
HyperionEO-3

HyspIRI
Ikonos
Ikonos-2
IRS (not specified)
IRS-Cartosat-1
IRS-Resourcesat-1
IRS-Resourcesat-2
JERS
Landsat (not specified)
Landsat-8
Landsat-ETM+
Landsat-TM
Landsat2
LandsatGLS
LandsatMSS
Lidar
LOJIC
MODIS
NAIP
NAPP
NGI-SouthAfrica
NOAA-AVHRR
PALSAR
Quickbird
Radarsat-1
Radarsat-2
RapidEye
RDAC
SPOT (not specified)
SPOT-1
SPOT-2
SPOT-4
SPOT-5
SPOT-6
SPOT5
TRMM
TRWIS
UAV
USGSDOQ
Worldview-2

Characteristics of remote sensing products

We also compiled information on the main type of remote sensing products being produced or used in the reviewed papers, i.e. as relating to climate, soil, spatial attributes, spectral identification (ID), image texture, topography, vegetation or water. Specific products mentioned by the authors of the papers, were found to be within the following list:

Aerosol Vapor Index (AVI)
Annual Insolation
Anthocyanin Reflectance Index (ARI)
Aspect
Atmospherically Resistant Vegetation Index (ARVI)
Biomass
Canopy Area Index
Canopy Water Content (CWC)
Carotenoid Reflectance Index (CRI)
Cellulose Absorption Index (CAI)
Chlorophyll Absorption in Reflectance Index (CARI)
Convexity
Corrected Transformed Vegetation Index (CTVI)
Difference Vegetation Index (DVI)
Digital Elevation Model (DEM)
Digital Terrain Model (DTM)
Distance
Elevation
Enhanced Vegetation Index (EVI)
Evapotranspiration Index (ET)
Forest Discrimination Index (FDI)
Global Environmental Monitoring Index (GEMI)
Green Cover Index (GCI)
Green Vegetation Index (GVI)
Green-Red Vegetation Index (GRVI)
Gross Primary Production (GPP)
Image Texture
Land Surface Albedo (LSA)
Land Surface temperature (LST)
Leaf Area Index (LAI)
Leaf N Concentration (LNC)
Light Use Efficiency (LUE)
Modified Simple Ratio Index (MSR)
MODIS Land Cover Product (LCP)

MODIS Land Cover Product (LCP)
MODIS Net Primary Production
NIR Plateau Index (NPI)
Normalized Difference Red Edge Index (NDRE)
Normalized Difference Soil Index (NDSI)
Normalized Difference Vegetation Index (NDVI)
Normalized Difference Water Index (NDWI)
Object Location (Photointerpretation)
Perpendicular Vegetation Index (PVI)
Photochemical Reflectance Index (PRI)
Ratio Vegetation Index (RVI)
Ratio-Vegetation-Index (RVI)
Red-Blue Ratio (RB)
Remote Digital Visual Inspection (RDVI)
Remotely Sensed Pigment Index (PI)
Simple-Ratio
Soil Brightness Index (BI)
Soil-Adjusted Total Vegetation Index (SATVI)
Soil-Adjusted Vegetation Index (SAVI)
Spatial Area
Spectrally Unmixed Soil Index
Standardized Blue-Green Ratio
Tasseled Cap Brightness Index
Tasseled Cap Greenness Index
Tasseled Cap Transformation Wetness
Temperature-Vegetation Wetness Index
Thiam's Transformed Vegetation Index (TTVI)
Topographic Wetness Index (TWI)
Total Vegetation Fractional Cover (TVFC)
Transformed Soil-Adjusted Vegetation Index (TSAVI)
Transformed Vegetation Index
Unspecified Spectral Reflectance Information
Vegetation Height
Visible-Near Infrared Vegetation Index (VNVI)
Water Band Index (WBI)
Weighted Difference Vegetation Index (WDVI)

Appendix D - Additional results and analyses from the literature review

This appendix provides additional information on temporal trends underlying our review classification. It also presents additional information discussed through the main text, regarding: the main characteristic of remote sensing sources (e.g. type of satellite sensor), data (e.g. hyperspectral, multispectral or radar information) and products (e.g., related to ecosystem properties namely water, soil, vegetation; see also Appendix C).

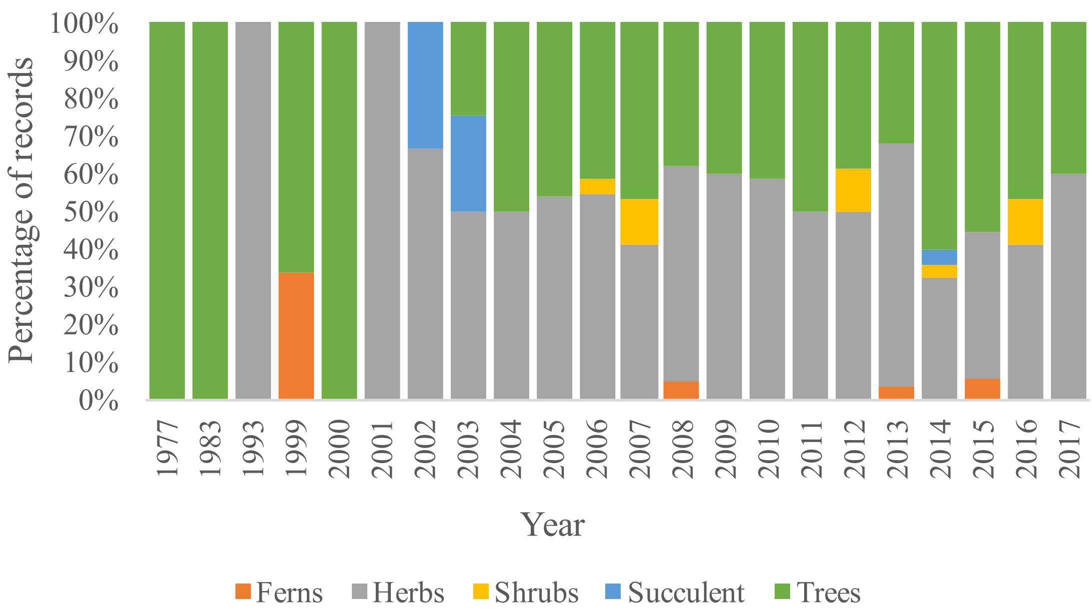


Figure S5.1. Temporal trends of the percentage of published record by growth form of the targeted invasive plant species. Our review shows that 49% of remote sensing applications deal with the management of herbaceous invaders (herbs), followed by trees (44%), shrubs (3%), ferns, and succulents (1.5% each). Pioneer studies concerned tree species, being gradually focusing as well on herb species through time.

Table S5.3. Number of observations across targeted species, as mentioned by the authors of the reviewed records. A diversification of targeted tree species was observed across our records, including the widest representativeness of studies on tree species such as *Prosopis glandulosa* and *Tamarix ramosissima.* A wide focus on herbaceous invasions is also observed, namely on *Spartina alterniflora*, *Eichhornia crassipes*, *Phragmites australis,* and *Leucaena leucocephala*.

Species	**Number of observations***
Several species	19
Tamarix spp. (unspecified)	16
Prosopis glandulosa	14
Spartina alterniflora	14
Tamarix ramosissima	14
Eichhornia crassipes	13
Phragmites australis	12

Leucaena leucocephala	10
Arundo donax	9
Lantana camara	9
Morella faya	8
Psidium cattleianum	8
Tamarix chinensis	8
Bromus tectorum	7
Lepidium latifolium	7
Euphorbia esula	5
Phragmites spp.	5
Carpobrotus edulis	4
Ligustrum lucidum	4
Lythrum salicaria	4
Acacia longifolia	3
Carduus nutans	3
Centaurea maculosa	3
Chromolaena odorata	3
Cirsium arvense	3
Cortaderia jubata	3
Elaeagnus angustifolia	3
Eragrostis lehmanniana	3
Eucalyptus spp.	3
Fallopia japonica	3
Linaria dalmatica	3
Lonicera maackii	3
Melaleuca quinquenervia	3
Pinus spp	3
Prosopis pallida	3
Prosopis velutina	3
Schinus terebinthifolius	3
Solanum mauritianum	3
Acacia spp.	2
Ambrosia artemisiifolia	2
Brassica tournefortii	2
Eucalyptus globulus	2
Eupatorium adenophorum	2
Fallopia sachalinensis	2
Fraxinus uhdei	2
Grevillea robusta	2
Gutierrezia sarothrae	2
Hakea spp.	2
Hedera helix	2

Heracleum mantegazzianum	2
Imperata cylindrica	2
Ligustrum spp.	2
Lonicera spp.	2
Miconia calvescens	2
Mimosa pigra	2
Myrica faya	2
Pennisetum ciliare	2
Pinus elliottii	2
Pittosporum undulatum	2
Psidium guajava	2
Pteridium aquilinum	2
Pteronia incana	2
Pueraria montana	2
Rubus armeniacus	2
Rubus moluccanus	2
Salvinia molesta	2
Tamarix parviflora	2
Trapa natans	2
Typha glauca	2
Typha spp.	2
Abies amabilis	1
Acacia caven	1
Acacia dealbata	1
Acacia mearnsii	1
Acacia melanoxylon	1
Acacia saligna	1
Ageratum conyzoides	1
Agropyron crsitatum	1
Ammophila arenaria	1
Ammophila breviligulata	1
Andropogon gayanus	1
Andropogon virginicus	1
Asclepias syriaca	1
Azolla filiculoides	1
Bromus rubens	1
Cannabis spp.	1
Carpobrotus acinaciformis	1
Casuarina equisetifolia	1
Cenchrus ciliaris	1
Cenchrus echinatus	1
Chrysanthemoides monilifera ssp rotundata	1

Clethra arborea	1
Cupressus lustitanica	1
Dendrocalamus spp.	1
Egeria densa	1
Elaeagnus umbellate	1
Elymus caput-medusae	1
Eragrostis curvula	1
Eupatorium spp.	1
Falcataria moluccana	1
Fallopia bohemica	1
Ficus rubiginosa	1
Flourensia cernua	1
Foeniculum vulgare	1
Frangula alnus	1
Gypsophila paniculata	1
Hedychium gardnerianum	1
Hovenia dulcis	1
Hydrocharis morsus-ranae	1
Hyptis suaveolens	1
Impatiens glandulifera	1
Juniperus ashei	1
Juniperus spp.	1
Lantana spp.	1
Larrea tridentata	1
Lemna obscura	1
Leptospermum spp.	1
Lespedeza cuneata	1
Leucosyris spinosa	1
Ligustrum sinense	1
Lynthrum salicaria	1
Megathyrsus maximus	1
Melinis repens	1
Mikania micrantha	1
Mimosa diplotricha	1
Ochlandra travancorica	1
Olea europaea subsp. africana	1
Oxalis pes-caprae	1
Parkinsonia aculeata	1
Parthenium hysterophorus	1
Parthenium spp.	1
Pennisetu ciliare	1
Pennisetum setaceum	1

Phleum pratense	1
Picea sitchensis	1
Pinus nigra	1
Pinus patula	1
Pinus radiata	1
Potentilla recta	1
Prosopis juliflora	1
Prunus serotina	1
Pteridium arachnoideum	1
Pteridium caudatum	1
Rhamnus cathartica	1
Rhododendron ponticum	1
Rosa rubiginosa	1
Rosa rugosa	1
Rubus fruticosus	1
Salix babylonica	1
Salix fragilis	1
Salix nigra	1
Salix spp.	1
Schismus arabicus	1
Schismus spp.	1
Scirpus mariquete	1
Senecio inaequidens	1
Senecio madagascariensis	1
Sisymbrium altissimum	1
Solidago altissima	1
Solidago canadensis	1
Solidago gigantea	1
Sorghum halepense	1
Spathodea campanulata	1
Stachys byzanthina	1
Tamarix gallica	1
Triadica sebifera	1
Triadica sebiferum	1
Tsuga mertensiana	1
Vachellia nilotica	1
Zizania latifolia	1

*Note that different species could also be targeted by the same record, reason why here we present the number of observations (i.e., number of times a given species was targeted by one record), instead of the number of records.

Table S5.4. Number of observations across the different habitat types (following the habitat classification scheme from IUCN: http://www.iucnredlist.org/technical-documents/classification-schemes/habitats-classification-scheme-ver3). A wide range of invaded habitat types has been targeted, with forests receiving higher focus (33% of records). Shrublands (15%), grasslands (13%), arable lands, and pasturelands (6% each) were also among the most targeted habitats, followed by an ensemble of studies in estuaries, coastal dunes, freshwater systems, and marshlands.

Habitat types	**Number of observations***
Forests	95
Grasslands	45
Several	45
Wetlands-Shrub Dominated Wetlands	29
Shrublands	21
Artificial-Arable Land	19
Artificial-Pasturelands	18
Wetlands-Forest Wetlands	16
Wetlands-Several	15
Marine Coastal-Sand Dunes	11
Marine Neritic-Estuaries	10
Artificial-Plantations	4
Marine Intertidal-Salt Marshes	4
Savanna	2
Wetlands-Permanent Freshwater Lakes	2
Artificial-Urban areas	1
Bare soil	1
Coastal dunes	1
Desert	1
Fynbos	1
Marine Intertidal-Mangrove	1
Wetlands-Delta	1
Wetlands-Marshlands	1

*Note that different habitat types could also be targeted by the same record, reason why here we present the number of observations (i.e., number of times a given habitat was targeted by one record), instead of the number of records.

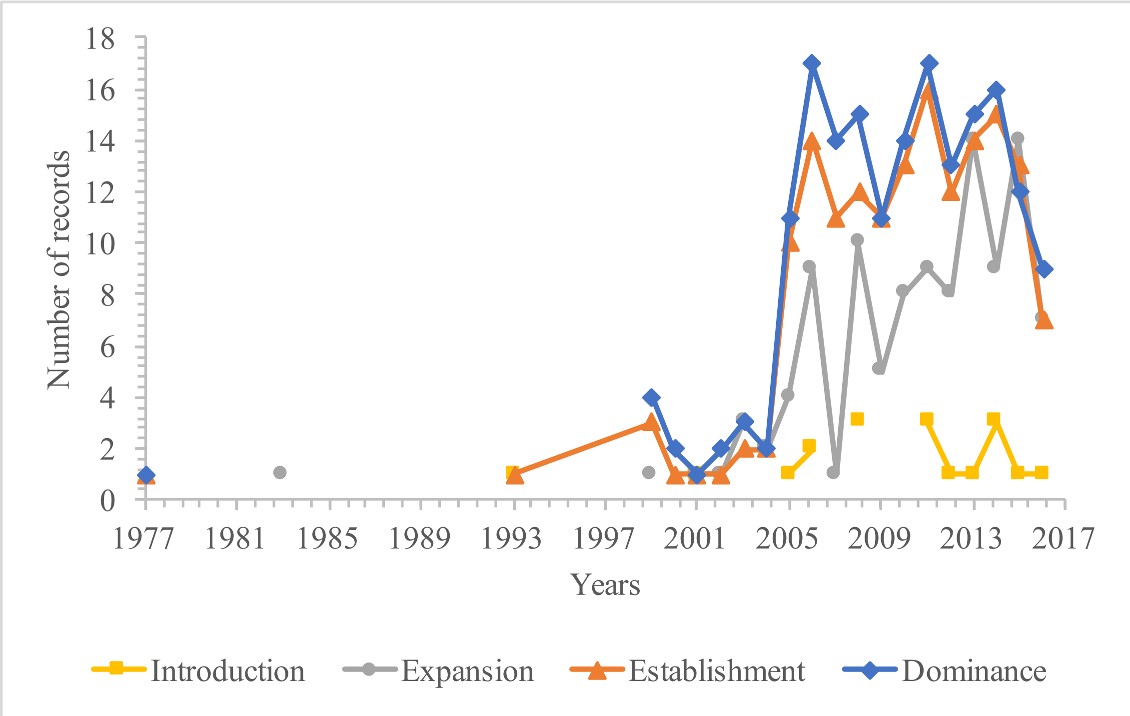


Figure 5.2. Temporal trends of published record by stages of the invasion process. Most remote sensing studies focused on later invasion stages, namely dominance (37% of all records) and establishment (33%). Interest on managing early invasion stages with remote sensing, i.e. expansion (23%) and introduction (4%), was only observed in the last decade.

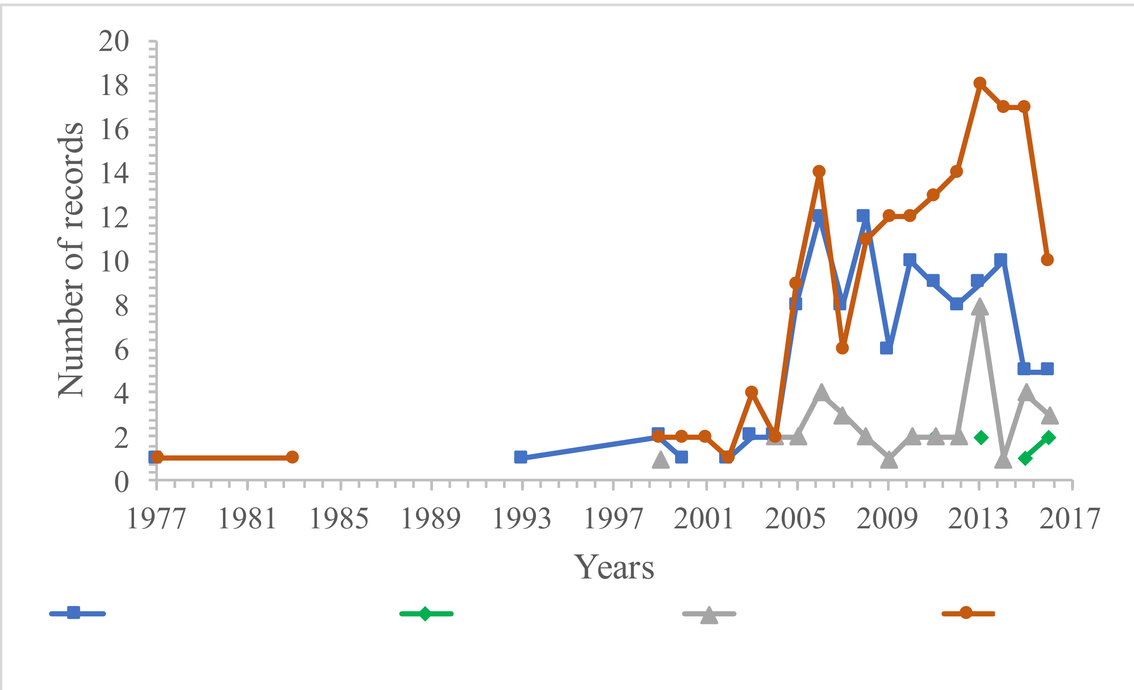


Figure 5.3. Temporal trends of published record by sources of remote sensing data. Overall, satellite derived products prevail in invasion management, making up 52% of all studies, followed by airborne (34%), groundthruth assessments (11%) and UAV-borne (2%). The figure also shows the emergence of improved satellite remote sensing data since the mid-2000s (until which airborne products dominated the field), and of high-resolution UAV-borne products during the last years.

Table S5.5. Number of observations across the different types of sources of remote sensing data, as mentioned by the authors of the reviewed records. A large proportion of studies that include satellite sources rely on data derived from Landsat TM/ETM+ sensors (21% of all records), followed by a minor proportion from MODIS (7%), Hyperion-EO (5%), Quickbird, and Ikonos (4%).

Source Type	**Number of observations***
Landsat-TM	57
Aerial Photos	43
Field Spectrometers	38
Landsat-ETM+	38
MODIS	30
Lidar	22
HyperionEO-1	18
Quickbird	18
Ikonos	17
AISA	15
AVIRIS	15
Worldview-2	12
HyMap	11
CASI	8
Landsat-8	8
SPOT (not specified)	8
NAIP	7
Aerial Videos	6
UAV photographs (unspecified)	6
GeoEye1	5
NOAA-AVHRR	5
USGSDOQ	5
ALIEO-1	4
AToMS	3
IRS-Resourcesat-1	3
PALSAR	3
COSMO-SkyMed-SAR	2
Huanjing-1	2
Landsat-GLS	2
Landsat-MSS	2
Radarsat-1	2
SPOT-2	2
ALS	1
APHS	1
Aster	1
CBRES-Ziuan-1	1
CBRES-Ziuan-2	1
CBRES-Ziuan-3	1
ENVISAT-ASAR	1

Formosat-2	1
GEOSAT	1
HJ-A	1
Hyperion (not specified)	1
HyperionEO-2	1
HyperionEO-3	1
HyspIRI	1
Ikonos-2	1
IRS (not specified)	1
IRS-Cartosat-1	1
IRS-Resourcesat-2	1
JERS	1
Landsat (not specified)	1
Landsat2	1
LOJIC	1
NAPP	1
NGI-SouthAfrica	1
Radarsat-2	1
RapidEye	1
RDAC	1
SPOT-1	1
SPOT-4	1
SPOT-5	1
SPOT-6	1
SPOT5	1
TRMM	1
TRWIS	1

*Note that different types of sources of remote sensing data could also be targeted by the same record, reason why here we present the number of observations (i.e., number of times a given habitat was targeted by one record), instead of the number of records.

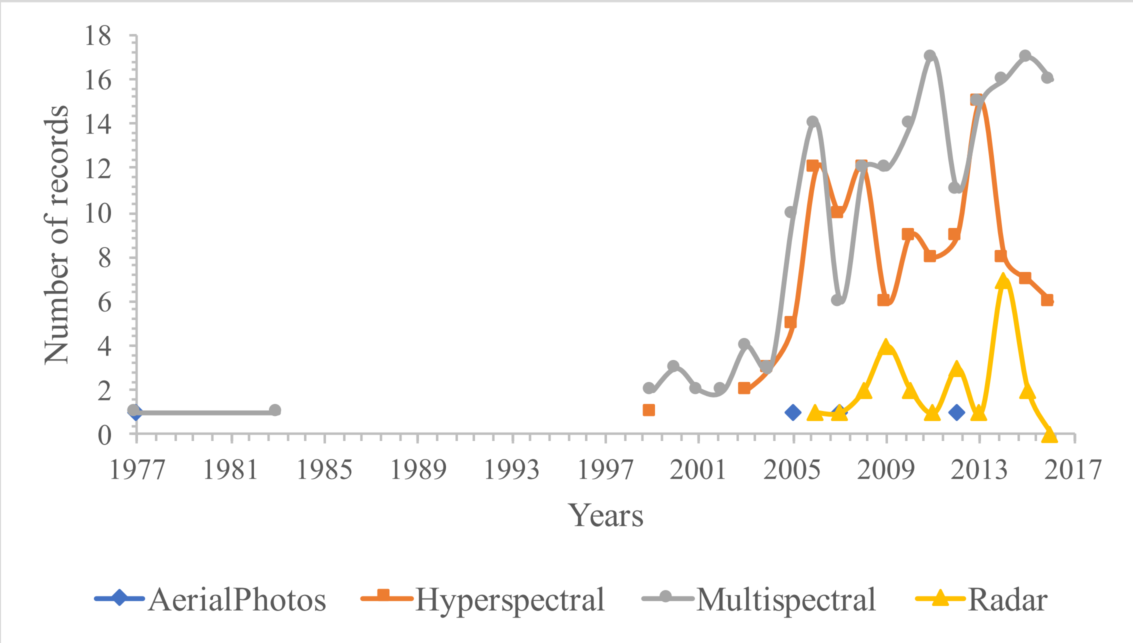


Figure S5.4. Temporal trends of published record by type of remote sensing data. The figure shows that despite the usage of multispectral data since the early 2000s, hyperspectral and radar information only grew during the last 10 years.

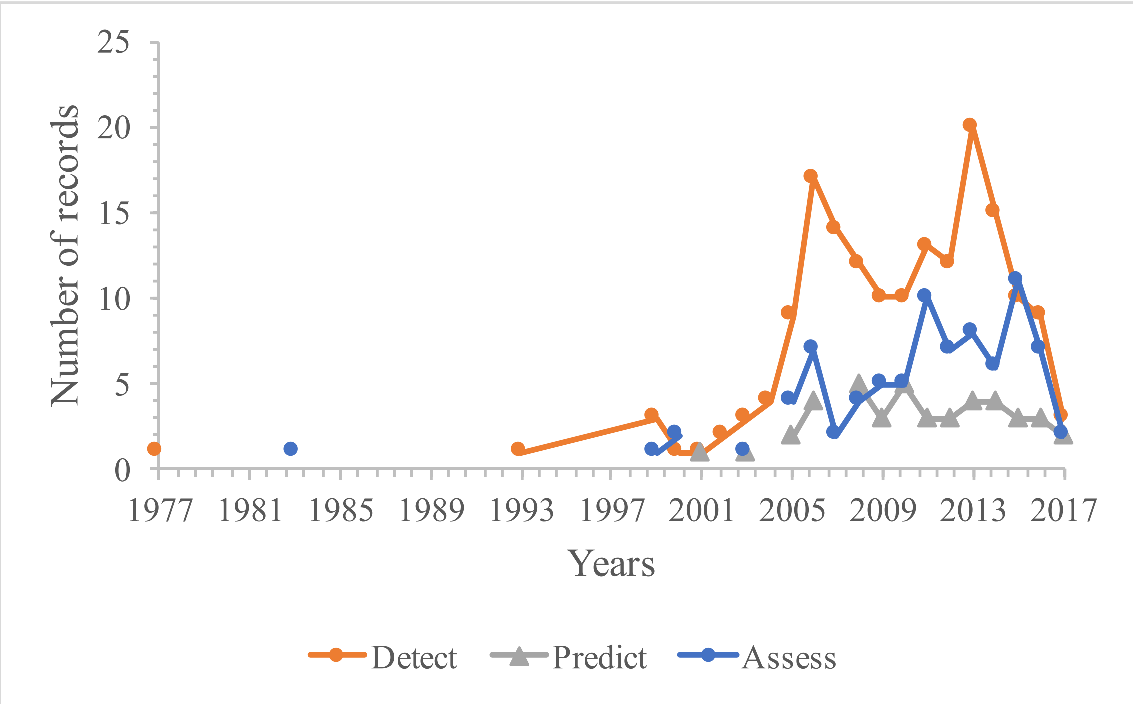


Figure S5.5. Temporal trends of published record by type of contribution form remote sensing to invasion management. Most studies focused on detecting plant invasions. The use of remote sensing to predict invasions has only emerged since the last 15 years. More recently, the assessment capacity of remote sensing became prevalent in the management of plant invasions.

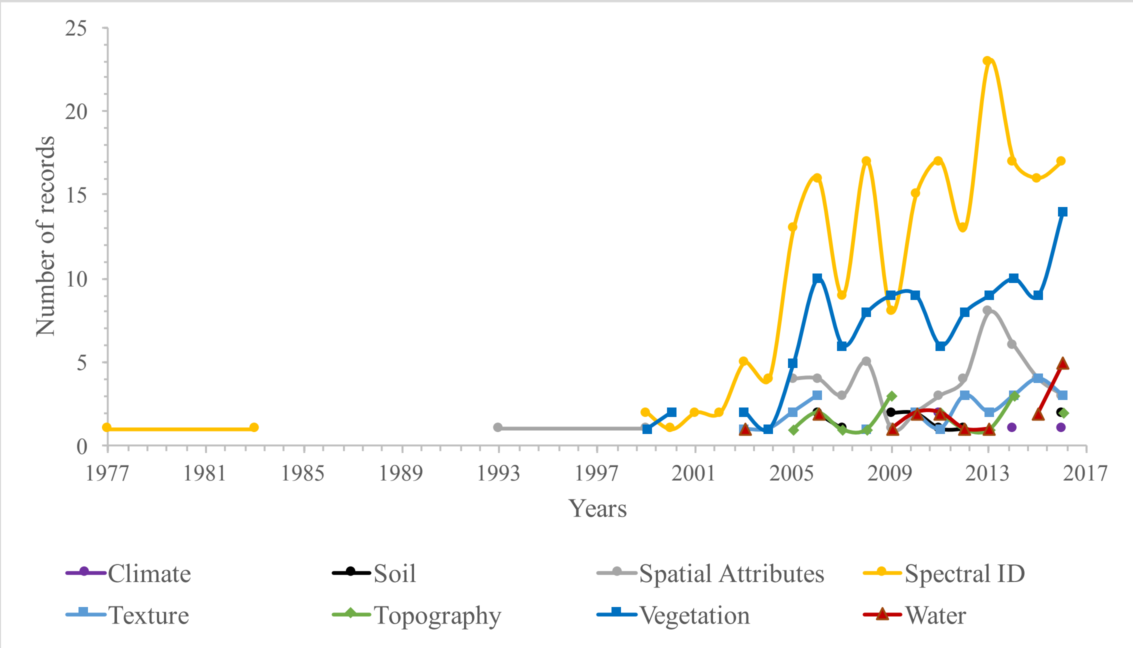


Figure S5.6. Temporal trends of published record by type of remote sensing products. The management of invasions has evolved from just using raw spectral and land-cover information to multiple remote sensing products dealing with vegetation (with 68% of the records), water (12%), topography (10%), soil (7%), and climate (3%) attributes.

Table S5.6. Number of observations across the different products of remote sensing data, as mentioned by the authors of the reviewed papers. Besides products dealing with the detection of invasive plants (e.g., photointerpretation, image texture, digital elevation model - DEM), the most used products have a biophysical meaning related to the carbon and energy dynamics, such as the NDVI (normalized difference vegetation index), EVI (enhanced vegetation index), vegetation height (radar output), NDWI (normalized difference water index), SAVI (soil-adjusted vegetation index), LAI (leaf area index), and LST (land surface temperature).

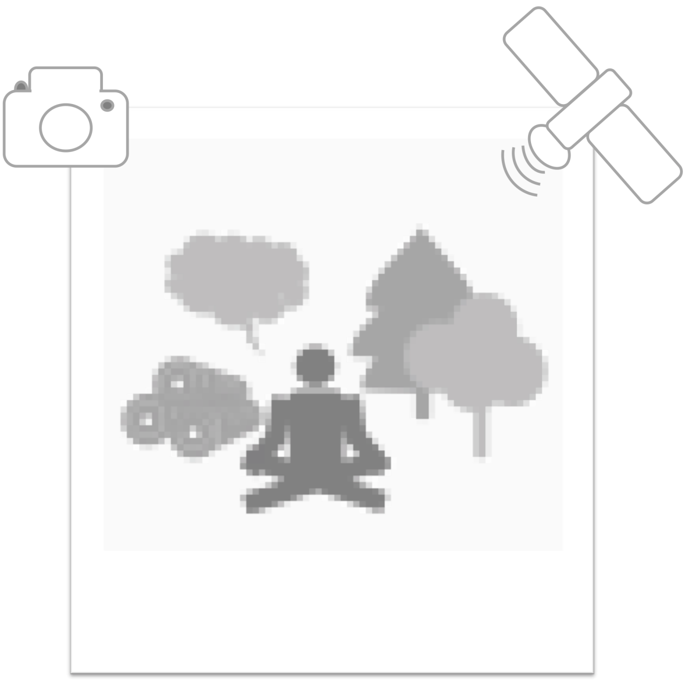
Remote Sensing Indices or Products	**Number of observations***
Unspecified Spectral Reflectance Information	207
Normalized Difference Vegetation Index (NDVI)	90
Object Location (Photointerpretation)	46
Image Texture	31
Vegetation Height	15
Digital Elevation Model (DEM)	14
Enhanced Vegetation Index (EVI)	12
Soil-Adjusted Vegetation Index (SAVI)	8
Elevation	8
Leaf Area Index (LAI)	7
Normalized Difference Water Index (NDWI)	6
Simple-Ratio	5
Ratio Vegetation Index (RVI)	4
Topographic Wetness Index (TWI)	3
Tasseled Cap Greenness Index	3
Green Vegetation Index (GVI)	3

Water Band Index (WBI)	2
Transformed Soil-Adjusted Vegetation Index (TSAVI)	2
Thiam's Transformed Vegetation Index (TTVI)	2
Spatial Area	2
Soil Brightness Index (BI)	2
Photochemical Reflectance Index (PRI)	2
Normalized Difference Red Edge Index (NDRE)	2
MODIS Land Cover Product (LCP)	2
Land Surface Albedo (LSA)	2
Cellulose absorption index (CAI)	2
Canopy Area Index	2
Biomass	2
Aspect	2
Annual Insolation	2
Aerosol Vapor Index (AVI)	2
Weighted Difference Vegetation Index (WDVI)	1
Visible Near-infrared Vegetation Index (VNVI)	1
Transformed Vegetation Index	1
Total Vegetation Fractional Cover (TVFC)	1
Temperature-Vegetation Wetness Index	1
Tasseled Cap Transformation Wetness	1
Tasseled Cap Brightness Index	1
Standardized Blue-Green Ratio	1
Spectrally Unmixed Soil Index	1
Soil-Adjusted Total Vegetation Index (SATVI)	1
Remotely Sensed Pigment Index (PI)	1
Remote Digital Visual Inspection (RDVI)	1
Red-Blue Ratio (RB)	1
Perpendicular Vegetation Index (PVI)	1
Normalized Difference Soil Index (NDSI)	1
Near infrared Plateau Index (NPI)	1
MODIS Net Primary Production	1
Modified Simple Ratio Index (MSR)	1
Light Use Efficiency (LUE)	1
Leaf N Concentration (LNC)	1
Land Surface Temperature (LST)	1
Gross Primary Production (GPP)	1
Green-Red Vegetation Index (GRVI)	1
Green Cover Index (GCI)	1
Global Environmental Monitoring Index (GEMI)	1
Forest Discrimination Index (FDI)	1
Evapotranspiration Index (ET)	1

Distance	1
Digital Terrain Model (DTM)	1
Difference Vegetation Index (DVI)	1
Corrected Transformed Vegetation Index (CTVI)	1
Convexity	1
Chlorophyll Absorption in Reflectance Index (CARI)	1
Carotenoid Reflectance Index (CRI)	1
Canopy Water Content (CWC)	1
Atmospherically Resistant Vegetation Index (ARVI)	1
Anthocyanin Reflectance Index (ARI)	1

*Note that different products of remote sensing data could also be targeted by the same record, reason why here we present the number of observations (i.e., number of times a given habitat was targeted by one record), instead of the number of records.

CHAPTER 6. REMOTE SENSING ANALYSIS OF CULTURAL ECOSYSTEM SERVICES



DISCLAIMER

This chapter is an original contribution of this thesis, submitted as an original paper and currently under review in journal *Remote Sensing of Environment* under the title "*Earth observation and social media: evaluating the spatiotemporal contribution of non-native trees to cultural ecosystem services*". The paper had the contribution of the following authors:

Ana Sofia Vaz[a,b,*], João F. Gonçalves[a], Paulo Pereira[a], Frederico Santarém[a,b], Joana R. Vicente[a,b,c], João P. Honrado[a]

[a]Research Network in Biodiversity and Evolutionary Biology, Research Centre in Biodiversity and Genetic Resources (InBIO-CIBIO), Universidade do Porto, Campus Agrário de Vairão, Rua Padre Armando Quintas, PT4485-661 Vairão, Portugal;
[b]Faculdade de Ciências, Universidade do Porto, Rua do Campo Alegre, s/n, PT4169-007 Porto, Portugal;
[c]Laboratory of Applied Ecology, CITAB − Centre for the Research and Technology of Agro-Environment and Biological Sciences, University of Trás-os-Montes e Alto Douro, Vila Real, Portugal.

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ABSTRACT

Understanding how the benefits from ecosystems, widely known as ecosystem services, are shaped by non-native biota is paramount to guide conservation management and achieve Sustainable Development Goals. Here, we developed a multidisciplinary approach to assess the spatiotemporal contributions of non-native trees to cultural ecosystem services. These contributions were interpreted from 1748 georeferenced social media photographs and evaluated against groups of predictors expressing environmental context (accessibility and wilderness; computed from Earth observation and ancillary spatial data) and landscape visual-sensory attributes (spectral diversity, colour and functioning; computed from MODIS and Sentinel-2 products). Contributions were analysed for each meteorological season, considering a multimodel inference framework applied to a National Park in Portugal. Overall, during Autumn, contributions of non-native trees to cultural services were higher than those from native trees, especially in colourful landscapes. During Spring, their contributions were promoted in accessible and homogeneous landscapes. Contrastingly, contributions from native trees prevailed over non-natives during Winter, particularly in remote areas. During Summer, non-native and native trees offered similar cultural contributions. These results are congruent with the phenology of prevailing tree species: deciduous natives occurring with coniferous non-natives and evergreen invaders, leading to colourful landscapes in Autumn, versus the dominance of blooming invaders in accessible areas during Spring. Results also match the seasonal dynamics of tourism demand in the National Park: the pursuit of wilder areas for ecotourism in Winter, versus the experience of popular recreational activities in Summer. We provide considerations for the adaptive management of non-native, often invasive, trees and discuss the usefulness of Earth observations for the research of cultural services.

Keywords: Multimodel inference; Plant invasion; Protected area; Recreation; Remote sensing; Tourism

6.1. Introduction

Nature contributes to human well-being by providing material benefits from ecosystems, which include provisioning (e.g. timber and food) and regulating (e.g. hazard mitigation and pollination) ecosystem services (MA, 2005). Ecosystems also offer non-material benefits, known as cultural ecosystem services, namely through touristic, recreational and aesthetic experiences (Fish et al., 2016). Evaluating cultural services is relevant for biodiversity protection (Nuñez and Simberloff, 2005); land tenure and management (Plieninger and Bieling, 2012); tourism and recreational revenues (Schirpke et al., 2018); and human heritage, identity and traditions (Ballet et al., 2018). Yet, cultural services can be intangible and hence they are frequently disregarded in research and decision-making (Blicharska et al., 2017).

Evaluations of cultural services include the use of public polls, which are often expensive and have limited spatial and temporal coverage (Wood et al., 2013); monetary appraisals, for which economic assumptions may fail (Chan et al., 2011); and biodiversity mapping, that tends to merely focus on the potential of cultural services (Spangenberg et al., 2014). In the “information age”, the use of big data from social media has become a promising approach to monitor human activities, as well as cultural preferences and perceptions towards nature, namely through the assessment of photographs shared via online networking (Figueroa-Alfaro and Tang, 2017; Nahuelhual et al., 2013; Van Berkel et al., 2018; Vaz et al., 2018a).

Understanding how cultural services are shaped by fingerprints of the Anthropocene, such as non-native biota (i.e. species that were introduced by humans to new geographic areas; Richardson et al., 2011) is paramount to achieve sustainable management (Kueffer 2017). This is particularly relevant for non-native trees which have been introduced to obtain resources worldwide (e.g. wood and ornamental features; Brundu and Richardson, 2015; Vaz et al., 2018a). Yet, tree species are within the most challenging non-native biota, particularly when spreading and becoming invasive in introduced regions, often disrupting the supply of provisioning and regulating services (e.g. water supply; Brundu and Richardson, 2015; Vaz et al., 2017a). However, how non-native trees contribute to cultural services is still a matter requiring attention (Kueffer and Kull, 2017; Vaz et al., 2018a; Vilà and Hulme, 2017).

Non-native trees can change visual-sensory landscape qualities (after Van Berkel et al., 2018), influencing people’s perception of cultural services (e.g. *“a beautiful tree”* or *“a scary tree*”; Shackleton et al., 2018; Vaz et al., 2017a). For instance, non-native trees can contribute to landscape homogeneity (e.g. large plantations or invasions), colour (e.g. through their “out-of-normal" and colourful flowers or leaves) and productivity (e.g. fast-growing species; Kueffer and Kull, 2017). Also, the environmental context (e.g. remoteness) determines people's accessibility to cultural services (Nahuelhual et al., 2013; Tenerelli et al., 2016; Wood et al., 2013), and hence perception of changes triggered by non-native trees (Shackleton et al., 2018). The contribution of non-native trees to cultural services further depends on the phenology of tree species (Kueffer and Kull, 2017; Shackleton et al., 2018), thus differing in space (e.g. confined or widespread trees) and between time seasons (e.g. deciduous or evergreen trees).

Although the cultural contribution of non-native trees has been discussed across space (Dickie et al., 2014; Kull et al., 2011; Nuñez and Simberloff, 2005; Vaz et al., 2018a), seasonal assessments are lacking. Earth observations can be very useful in this regard (Krishnaswamy et al., 2009; Van Berkel et al., 2018; Vaz et al., 2018b; Vaz and Santos, 2018). For instance, freely-available imagery from satellite sensors (e.g. MODIS and Sentinel-2) can capture landscape patterns and attributes that sustain the provision of ecosystem services (Alcaraz-Segura et al., 2013; de Araujo Barbosa et al., 2015; Van Berkel et al., 2018). These data can be acquired for different seasons according to plant phenology, particularly when using open-source platforms with high processing ability (e.g. Google Earth Engine; Gorelick et al., 2017; Kwok, 2018).

Evaluating the seasonal contribution of non-native trees can aid decision-makers and managers in understanding people's tolerances, perceptions and preferences, as well as in identifying risks and opportunities for sustainable development (Dickie et al., 2014; Shackleton et al., 2018; Vaz et al., 2017b). By combining in-field photographs from social media platforms with satellite Earth observation and ancillary spatial data, here we analyse how non-native trees contribute to cultural ecosystem services in space and across annual seasons. We further evaluate how these contributions relate to the environmental context (i.e. accessibility and wilderness) and to visual-sensory features of the landscape (i.e. spectral diversity, colour, and functioning). Our approach is tested in a National Park of Portugal, where informed management regarding non-native trees is needed to safeguard nature values and promote cultural benefits. Finally, we discuss the implications of our results for the management of non-native trees and outline opportunities for advancing the research of cultural ecosystem services through Earth observations.

6.2. Methods

6.2.1. Workflow

Our workflow included three main steps: (A) data collection and processing, (B) spatiotemporal analysis, and (C) multimodel inference (Figure 6.1). First, we compiled a georeferenced dataset of in-field photographs for our test area, from social media; and computed a set of predictors potentially explaining the contribution of non-native trees to cultural services, from satellite Earth observation and ancillary data (see section 6.2.3). Then, the photographic dataset was analysed to evaluate the spatiotemporal contributions of non-native trees to cultural services in the test area, across the four meteorological seasons: Winter (December-February), Spring (March-May), Summer (June-August) and Autumn (September-November; section 6.2.4). Finally, non-native tree contributions were used as dependent variables in a multimodel inference framework intended to evaluate the explanatory power of the set of predictors (related to environmental context, landscape visual-sensory, or both; section 6.2.5).

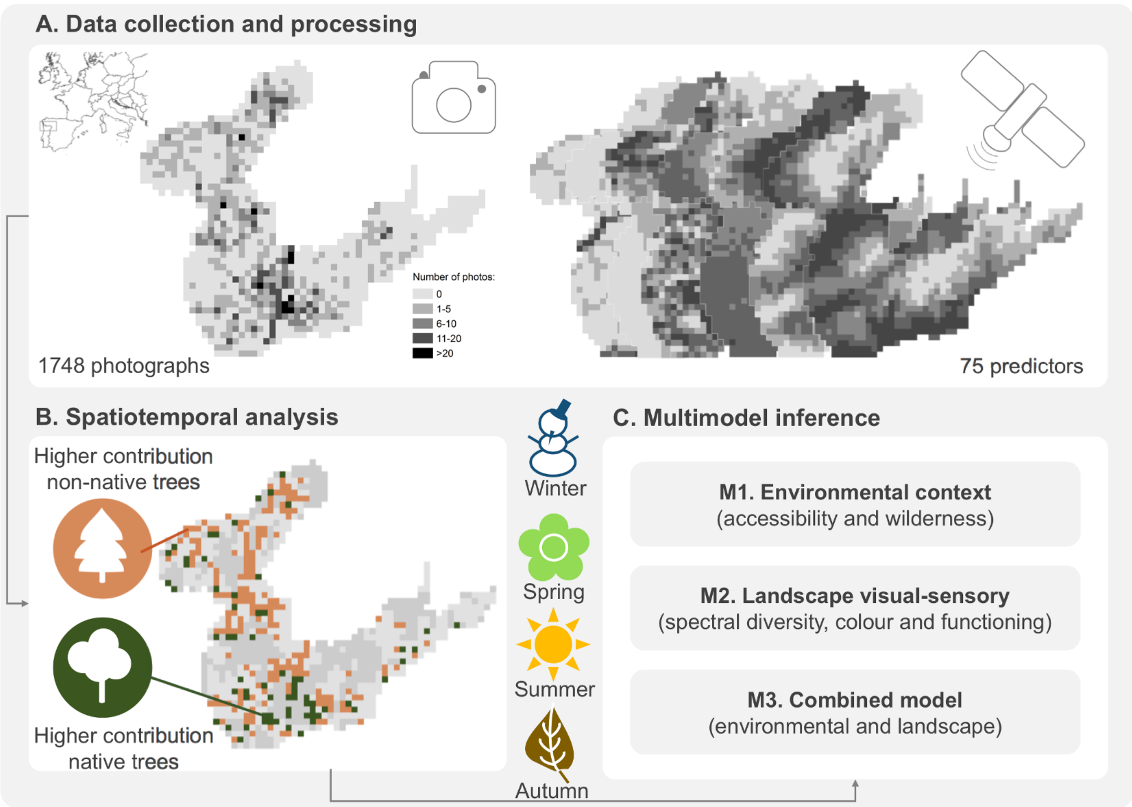

Figure 6.1. The workflow considered to assess the contribution of non-native trees to cultural services: (A) collection of social media photographs, and computation of Earth observation predictors; (B) calculation of the contribution of non-native trees to cultural services in the test area and across meteorological seasons; (C) multimodel inference, testing which set of predictors best explains the spatiotemporal contributions of non-native trees to cultural services.

6.2.2. Test area

The test area (950 km^2) is located in the northwest of Portugal (41º 41' N, 8º 25' W; Figure 6.1). It includes the only National Park in the country ("Peneda-Gerês", established in 1971) and also a Special Protected Area (SPA) and a Site of Community Importance (SIC; EU Natura 2000). The climate is Temperate to sub-Mediterranean, with mean annual temperature between 13 and 15 °C and annual rainfall usually over 2000 mm. Elevation ranges between 100 and 1548 m and the prevailing bedrock type is granite (Honrado, 2003). The area comprises biodiversity-rich mountain landscapes with native scrublands, grasslands and *Quercus* woodlands (Honrado, 2003; Vaz et al., 2015). During the 19th century and before its establishment as a protected area, many non-native trees were introduced, including currently widespread and invasive *Acacia* species (Fernandes, 2008). The area holds a rich archaeological (e.g. megalithic monuments and signs of Roman occupation) and historical heritage (e.g. traditional celebrations and land-use practices; Santarém et al., 2015). However, since agro-pastoral and forestry activities suffered from rural abandonment, recreational and touristic activities with high socio-economic potential have been promoted (Kastenholz and de Almeida, 2008; Santarém et al., 2015). Thus, management actions in the test area need to be guided by solid strategies that safeguard biodiversity alongside with cultural benefits.

6.2.3. Data collection and processing

Cover of native and non-native trees

We applied a regular grid of 1 km^2 to the test area (total of 1008 cells). For each grid cell we collected information on the cover area of non-native and native trees (based on Honrado, 2003; Vaz et al., 2018a). Cover areas were obtained from the detailed and freely available Land Cover Map (COS 2007; at http://mapas.dgterritorio.pt/), complemented with information from the National Forest Inventory (at http://www2.icnf.pt/portal/florestas/ifn/ifn6; Appendix A).

In-field photographs from social media

Cultural services were evaluated through the screening of photographs from two popular social media platforms: Flickr and Wikiloc (following Casalegno et al., 2013; Figueroa-Alfaro and Tang, 2017; Nahuelhual et al., 2013; Vaz et al., 2018a). We collected georeferenced photographs prior to 2018 and within the borders of the test area. Flickr data were collected using the Application Programming Interface (API) together with data collection tools which we developed with Python 3.5 (Appendix B). Wikiloc data was extracted from Google Earth (https://www.google.com/earth/). We registered the spatial location (latitude and longitude) and date (month and year) of each photograph. We then classified each photograph according to the dominant non-native or native tree species (following Vaz et al., 2018a), and based on its main focus (see Table 6.1; following Hausmann et al., 2017; Oteros-Rozas et al., 2017). Details on tree taxonomy (i.e. genera or species name) and physiology (e.g. bloomed trees) were recorded. We excluded photographs with irrelevant subjects (e.g. advertisements, pamphlets and drawings). Photographs protected by users' privacy were not downloaded nor analysed.

Table 6.1. Categories considered to classify each photograph according to its main focus.

Category	Description
Posing	People looking at the camera, with recognisable faces
Landscape	Pictures showing wide views of an area, with visible horizon
Species	Trees or parts of trees (e.g. flowers or leaves) as main subject
Human structures	Pictures showing human infrastructures (e.g. houses or monuments)
Human activities	People engaged in recreational activities (e.g. hiking and biking), including related objects (e.g. canoes and bicycles)

Predictors

Based on previous research and data availability, we considered 75 variables as candidate predictors of the contribution of non-native trees to cultural services (Table 6.2; Appendix C). Predictors were arranged into two broad groups: (1) environmental context - as area accessibility and wilderness derived from topographic information, hydrographic and road networks, surveillance effort and visual accessibility (Guerra et al., 2013; Nahuelhual et al., 2013; Tenerelli et al., 2016; Vicente et al., 2016); and (2) landscape visual-sensory attributes - as landscape spectral heterogeneity, colour diversity and functioning, calculated from satellite information for each meteorological season (Alcaraz-Segura et al., 2013; Cabello et al., 2012; Krishnaswamy et al., 2009; Van Berkel et al., 2018). Predictors were tested for pair-wise correlations (Pearson correlation) and multicollinearity (VIF: variance inflation factor). Predictors with correlations > 0.6 and VIFs > 4 were no longer considered (Fox and Weisberg, 2011; see Appendix C for details).

Table 6.2. Groups of variables considered as candidate predictors of non-native tree contributions to cultural services.

Environmental context (accessibility and wilderness)	
Topography	Information on elevation, slope and aspect (http://biodiversidade.eu/)
Visual accessibility	Viewshed dimension expressing the number of pixels at 30 m from the centroid of each cell (ASTER GDEM)
Accessibility effort	Remoteness index based on information related to e.g. rough terrain or distance to roads
Road and river density	Local density of roads and rivers in each cell, based on road and hydrographic networks (http://biodiversidade.eu/)
Landscape visual-sensory (spectral heterogeneity, colour and functioning)	
Landscape heterogeneity	Number and diversity (Shannon and Simpson) of clusters, retrieved from unsupervised classification of satellite imagery (Sentinel-2: 10 m spatial resolution, 10-days temporal resolution, years 2015-2017)
Colour diversity	Mean and standard-deviation of reflectance values in each cell, retrieved from satellite imagery (Sentinel-2: 10 m spatial resolution, 10-days temporal resolution, years 2015-2017) focusing on the RGB (Red-Green-Blue) visible part of the electromagnetic spectrum)
Vegetation functioning	Vegetation productivity, as the mean and standard deviation of the Enhanced Vegetation Index (EVI) in each cell obtained from satellite imagery time-series of Ecosystem Functional Attributes (MOD13Q1 product from MODIS sensor, at 250 m resolution, 16-days temporal resolution, years 2011-2017)

6.2.4. Data analyses: evaluating non-native tree contributions to cultural services

Prior to analysing the contribution of non-native trees to cultural services, chi-square tests were considered to evaluate the significance of associations among the date (year and season), focus (Table 6.1), and type of dominant tree (non-native or native) across photographs. To visualise the associations among the previous categories, we applied a multidimensional unfolding (MDU) based on matrices of preference data (see Appendix F for details).

Non-native tree contributions were evaluated through the odds ratio (Borenstein et al., 2009). The odds ratio was computed as the number of photographs dominated by non-native or native trees (i.e. observed contribution of non-natives in the photograph) per meteorological season, weighted by the proportion of cover area of non-native and native trees in each grid cell (i.e. the expected contribution of non-natives and natives in the area). We calculated the weighted odds ratio (wOR) using Peto's method and the DerSimonian-Laird random effects model under non-parametric permutation tests, with 1000 iterations (following Borenstein et al., 2009; Vaz et al., 2018a). Weighted odd ratios higher or lower than zero express a higher or lower contribution of non-native trees to the cultural service compared to native trees, respectively. Odds ratios equal to zero indicate similar contributions between non-native and native trees, and thus no preference for non-native (or native) trees (Vaz et al., 2018a; see Appendix D for details).

6.2.5. Multimodel inference: explaining non-native tree contributions to cultural services

Three competing models (M1-M3) were considered in a multimodel inference framework (Burnham and Anderson, 2003) to test whether the contribution of non-native trees to cultural services was mostly explained by: M1 - predictors expressing the environmental context; M2 - predictors expressing seasonal visual-sensory landscape features; or M3 - a combination of both (Figure 6.1C; Table 6.1). Given the meta-analytical nature of our response variable (expressed by the odds ratio), we implemented random-effect meta-regression models, using the maximum likelihood estimation using R (*glmulti* package; Calcagno, 2013). For model comparison, we calculated the Akaike Information Criterion difference (ΔAICc), as ΔAICc = AICc initial - AICc minimum (where AICc initial is the second-order AICc of the competing model; AICc minimum is the second-order AIC of the best model in the set). We further considered the weight (w_i) of each competing model, that represents the proportion of evidence from a competing model in relation to the total evidence from all models (ranging between 0 and 1). We used the Nagelkerke's deviance D2 (based on null-model testing) and Pearson correlation R2 (between predicted and observed values) as goodness-of-fit measures (Burnham and Anderson, 2003; Dormann et al., 2018).

6.3. RESULTS

6.3.1. Overview of the photographic dataset

A total of 7227 photographs was retrieved from Flickr (44%) and Wikiloc (66%), from which 1748 photographs were subsequently considered (see Appendix B). The date of the photographs spanned from 2003 to 2017. The number of photographs was fairly similar across seasons: Winter (24.02 %), Summer (24.17 %), Spring (22.96 %), Autumn (28.86 %). Most photographs focused on landscapes (50.96 %), followed by human structures (28.66 %) and activities (8.68 %), species (7.92 %) and finally posing (3.79 %; Figure 6.2). The most photographed non-native tree species were *Pinus pinaster* (53.23 %), *Pseudotsuga menziesii* (8.48 %), *Chamaecyparis lawsoniana* (8.13 %), *Acacia dealbata* (7.51 %), *A. melanoxylon* (6.98 %) and *Eucalyptus globulus* (4.33 %; Appendix E).

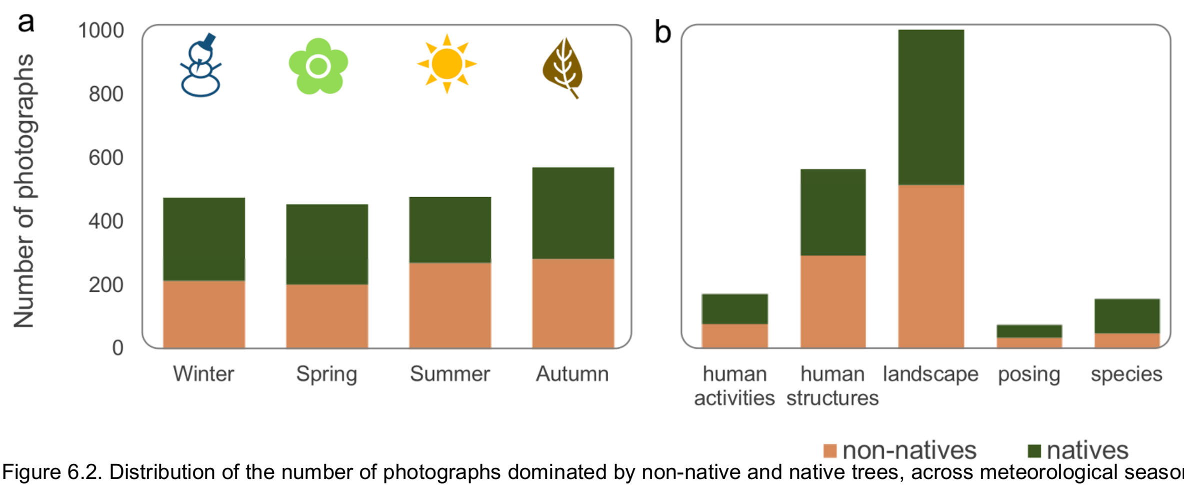

Figure 6.2. Distribution of the number of photographs dominated by non-native and native trees, across meteorological seasons (a) and in relation to the photograph focus (b).

Chi-square tests revealed significant associations between the dominant tree (non-native and native) and the meteorological season (chi = 25.812; df = 6; $p < 0.001$). No significant associations were found with the year (chi = 30.59; df = 22; $p > 0.10$) nor with the photograph focus (chi = 70.99; df = 8; $p > 0.05$). The MDU (stress = 0.63) also did not show an evident pattern regarding the association among the former (see Appendix F for full results).

6.3.2. Spatiotemporal contributions of non-native trees to cultural ecosystem services

Non-native trees hold slightly higher contributions to cultural services than native trees, with 20 % and 18 % of all cells respectively showing positive and negative odds ratio values (Figure 6.3). The wOR was also significant and positive (wOR = 0.58; $p < 0.001$). The spatial projection of odds ratios showed the prevalence of positive values in western and central parts of the test area, whereas negative values prevailed in eastern and southern areas.

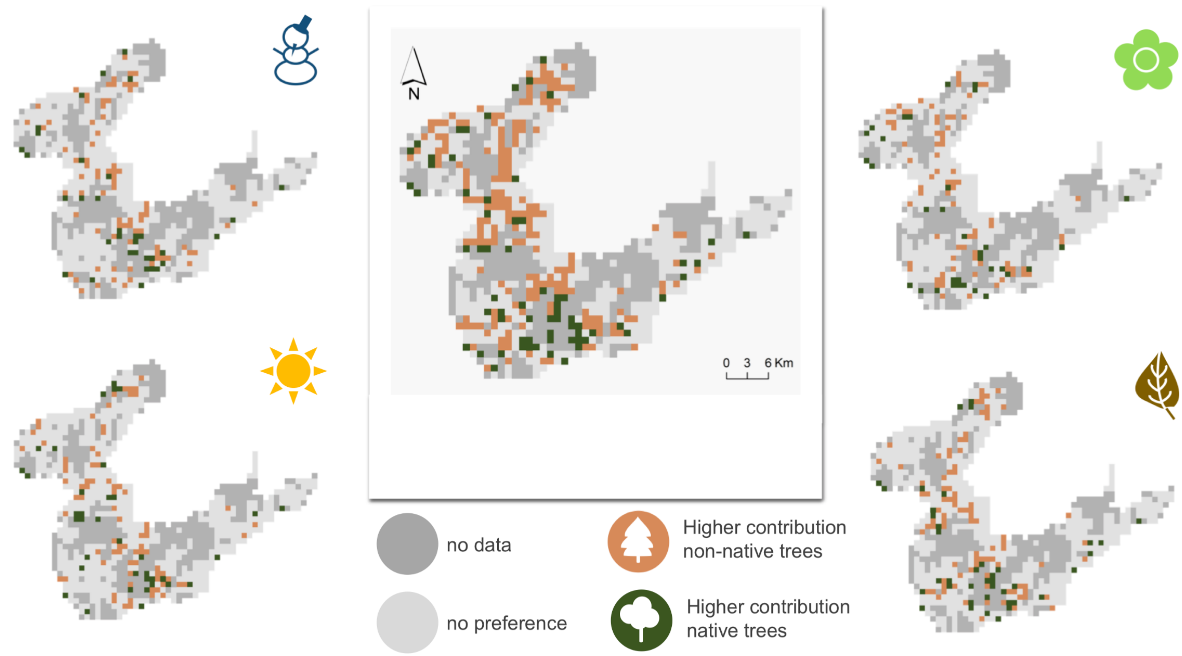

Figure 6.3. Spatial representation of the contribution of non-native and native trees to cultural ecosystem services, in general (centre) and for each meteorological season: Winter (upper-left), Spring (upper-right), Summer (bottom-left) and Autumn (bottom-right). No data refers to the absence of photographs.

Regarding seasonality, we found significant and positive wOR values for Autumn (wOR = 0.23; $p < 0.01$) and Spring (wOR = 0.19; $p < 0.01$), but negative wOR values for Winter (OR = -0.19, $p < 0.001$). For Summer, the wOR was non-significant (wOR = 0.02, $p > 0.5$; Appendix G).

6.3.3. Predictors of the contribution of non-native trees to cultural services

The contribution of non-native trees to cultural services was explained by predictors expressing both the environmental context and landscape visual-sensory features, as shown by the ranking of models based on AICc and w_i values (Table 6.3).

Table 6.3. Summary of results from the multimodel framework. Models are presented from the best to the least fit hypothesis, based on the Akaike Information Criterion difference (∆AICc) and weight (wi). D2: deviance; VIF: variance inflation factor; QM: heterogeneity of the explained response variable (tested by means of the Q statistics). Next to each predictor, we indicate whether the predictor was positively (+) or negatively (-) related to the contribution of non-native trees (Appendix H shows full results).

	∆AICc	**w_i**	**D2**	**VIF**	**QM**	**Predictors**
Winter						
M1	0.00	0.73	0.67	1.49	3.13	Elevation (+) Road density (-)
M3	0.10	0.21	0.40	1.54	4.71	Road density (-) Colour diversity (+) Elevation (+)
M2	1.32	0.06	0.11	1.49	1.34	Colour diversity (+)
Spring						
M3	0.00	0.68	0.59	1.23	4.08	Diversity of clusters (-) Road Density (+)
M2	0.10	0.23	0.36	1.08	3.63	Diversity of clusters (-) Colour diversity (+)
M1	1.26	0.09	0.32	1.26	3.30	Accessibility effort (-) Road Density (+)
Summer						
M3	0.00	0.47	0.25	1.24	2.57	Road Density (+) Colour diversity (-)
M1	0.10	0.34	0.15	-	1.57	Road Density (+)
M2	0.77	0.19	0.08	-	0.81	Colour diversity (-)
Autumn						
M3	0.00	0.62	0.54	1.27	5.84	River Density (-) Colour diversity (+)
M1	0.07	0.30	0.43	1.49	4.93	River Density (-) Elevation (+)
M2	1.10	0.08	0.34	1.21	3.84	Number of clusters (-) Colour diversity (+)

The best competing model for Autumn (M3: w_i = 0.62; QM = 5.84; $p < 0.05$) and Spring (M3: w_i = 0.68; QM = 4.08; $p < 0.05$) was based on a combination of predictors describing the environmental context and landscape visual-sensory features. For Winter, the best model was based on predictors related to accessibility (M1: w_i = 0.73; QM = 3.13; $p < 0.05$). For Summer, none of the considered models and predictors significantly explained the variation of non-native tree contributions ($p > 0.05$).

Regarding the predictors, in Autumn, non-native tree contribution was negatively related to river density (R2 = -0.10; $p < 0.05$), but positively related to colour diversity (standard deviation of the green band from Sentinel-2; R2 = 0.24; $p < 0.05$). In Spring, non-native tree contribution was negatively related to the diversity of landscape clusters (Shannon diversity; R2 = -0.35; $p < 0.05$), but positively to road density (R2 = 0.08; p = 0.05). In Winter, road density (R2 = -0.10; p = 0.05) also explained most of the contribution of non-native trees (negative relation), together with elevation, which hold a positive relation (R2 = 0.24; p = 0.05; Figure 6.4).

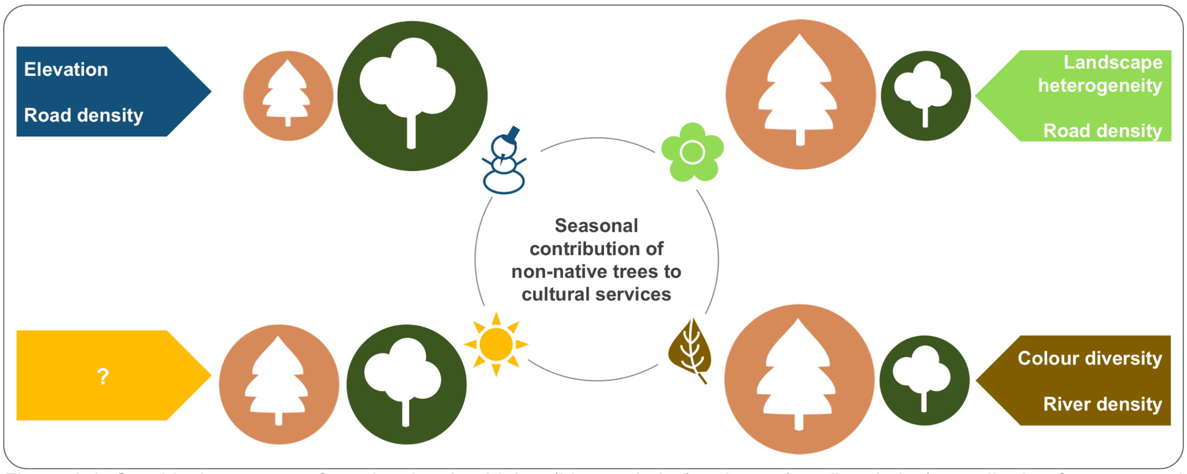

Figure 6.4. Graphical summary of results showing higher (bigger circles) or lower (smaller circles) contribution from non-native trees (orange circles) to cultural services, compared to native trees (green circles). The figure also shows the predictors that mostly explained the contribution of non-native trees in each meteorological season (arrows).

6.4. DISCUSSION

6.4.1. Spatiotemporal contributions of non-native trees to cultural ecosystem services

Our multidisciplinary approach revealed higher contributions from non-native trees (compared to native trees) to cultural ecosystem services in parts of the test area that are more accessible and prone to recreational activities. Contrastingly, native trees showed higher contributions in remote areas with high conservation value in the National Park. These patterns match the known distribution of non-native (particularly, invasive *Acacia* trees) and native trees (specifically, old-growth *Quercus* forests) in the protected area (Fernandes, 2008; Vicente et al., 2016). Our analysis also suggested no influence of species nativeness when photographing nature (in agreement to Oteros-Rozas et al., 2017; Vaz et al., 2018a); yet, there was a dominance of landscape-focused photographs. Landscapes change alongside dominant vegetation (Alcaraz-Segura et al., 2013; Van Berkel et al., 2018), which in our area corresponds mainly to woodlands of native deciduous trees (with seasonal changes in their spectral response), non-native conifers, and/or evergreen wattles (*Acacia* spp.) and eucalypts (*Eucalyptus* spp.; Honrado, 2003).

Our results showed that the contribution from these trees to cultural services changed across seasons. During Autumn there was a higher contribution of non-native trees. This was particularly evident in more colourful landscapes, likely due to the intermix between deciduous (native), conifer and evergreen (non-native) trees, particularly in accessible areas (Martínez Pastur et al., 2016; Vicente et al., 2016). Non-native trees were also preferred over natives during Spring, when they exhibit their most attractive “exotic” features (e.g. acacias’ yellow flowers and pines’ cones), making natives less conspicuous (Kueffer and Kull, 2017). In Winter, native trees contributed more to cultural services (especially in remote areas), whereas in Summer non-native and native trees showed similar contributions. These results match the seasonal dynamics in tourism demand, with a “specialised” public searching for ecotourism in native areas during Winter, and a generalist public interested in more popular recreational experiences (e.g. entertainment facilities) during Summer (Ceauşu et al., 2015; Kastenholz and de Almeida, 2008; Martínez Pastur et al., 2016; Santarém et al., 2015).

Considering these results, we draw some guidelines towards the achievement of Sustainable Development Goals and Targets (https://sustainabledevelopment.un.org; Wood et al. 2018).

Particularly, we recommend biosecurity efforts to prevent and mitigate non-native species' effects on natural and cultural heritage (including ecotourism) in the National Park (e.g. Goal 15; also Targets 8.9, 12.8, 11.4). Biosecurity efforts should include preventive actions through environmental education and in situ eradication and control of non-native trees (Marchante and Marchante, 2016; Reimer and Walter, 2013). Those efforts should prioritise sites with highest occurrence and cultural preference for non-native trees (also more prone to recreational tourism). Biosecurity efforts should have no considerable impact on tourism revenues, as no public preferences were shown during the peak season (i.e. Summer). Instead, they would be relevant for controlling widespread invasives (e.g. *Acacia dealbata* and *A. melanoxylon*), as well as for preventing the naturalisation and invasion by other non-natives (e.g. *Pseudotsuga menziesii* and *Robinia pseudoacacia*). Nevertheless, these efforts should consider the contributions that non-native trees might have on other ecosystem services (Vaz et al., 2017a). Failure to do so will likely hamper attempts to ensure that bundles of ecosystem services are included in current and future land planning (Casalegno et al., 2013).

6.4.2. Assessing cultural ecosystem services from Earth observations

The emergence of computational approaches in social sciences, namely through the evaluation of social media data, constitutes a revolutionary way to compile and analyse people's experiences and interactions with ecosystems (Dunkel, 2015; Lazer et al., 2009). This is relevant for advancing knowledge about nature's cultural benefits to people (Díaz et al., 2018), and hence cultural ecosystem services through an objective, quantifiable and spatiotemporal way (Dunkel, 2015; Figueroa-Alfaro and Tang, 2017). Concurrently, satellite remote sensing provides spatially- and temporally-explicit information on the prevalence of, and accessibility to, visual and sensorial characteristics of ecosystems (Alcaraz-Segura et al., 2013; Van Berkel et al., 2018). Beside the availability of high-resolution Earth observation data (e.g. Sentinel-2), there is also an investment in open-source and user-friendly platforms with increasing processing capacity (e.g. Google Earth Engine; Kwok 2018). We have shown that the combination of social media photographs and Earth observation data can be useful for the research of cultural ecosystem services and their determinants.

Still, some methodological considerations are recognised. The spatial reference precision of social media photographs can bias the geolocation of collected data (Figueroa-Alfaro and Tang, 2017). This bias was likely insignificant in our study, due to the aggregation of photographs at 1 km spatial resolution. Also, in a multimodel framework the predictive ability of a competing model is evaluated in relation to the other models, not necessarily meaning that the best model is able to explain the full range of variations. The process of cultural evaluation of ecosystems may differ across social-ecological contexts and (groups of) individuals (Kull et al., 2011; Shackleton et al., 2018; Vaz et al., 2018a). Therefore, in order to further understand cultural preferences towards non-native trees, future research should examine the motivations underlying choices and perceptions (Shackleton et al., 2018) in relation to other (social) determinants (e.g. socio-demography, economy; Tenerelli et al., 2016; van Zanten et al., 2016; Vaz et al., 2018a). Including complementary types and sources of information should be encouraged as platforms evolve and more data becomes available (Oteros-Rozas et al., 2017). As social data and high-resolution satellite information becomes publicly available, their use may be ethically sensitive, increasing scientific responsibility on the interpretation and communication of social patterns (Baumber et al., 2018; Dunkel, 2015; Van Berkel et al., 2018; reason why we did not retrieve or analyse data protected by user's privacy).

Nevertheless, our study considered the most relevant available data to assess the spatial and seasonal contributions of non-native trees to cultural ecosystem services. Our results suggested that the contribution of non-native trees to cultural services in the test area is defined by the local abundance and phenology of these species. Approaches based on Earth observation time-series can help to detect these species alongside their effects on recipient ecosystems (Alcaraz-Segura et al., 2013; Vaz et al., 2018b). Also, the cultural value of a natural feature depends on people's accessibility to that feature (Ceauşu et al., 2015; Figueroa-Alfaro and Tang, 2017; Reimer and Walter, 2013). For instance, road sides are prone to the occurrence of non-natives but also to the movement of people, improving the chance of a given non-native tree being culturally valued. In this sense, Earth observation data is useful to feed modelling frameworks that can predict the spatial patterns of these species, e.g. along dispersal corridors (Rocchini et al., 2015).

Finally, the cultural value of nature depends on visually-attractive characteristics which succeed in capturing human attention (Van Berkel et al., 2018). This is particularly relevant for non-native trees, which often exhibit higher growth performance and "out-of-normal" phenological traits, particularly when natives are leafless and less conspicuous (Shackleton et al., 2018; Vaz et al., 2018a). Satellite data can aid in these assessments, capturing information on ecosystem functioning and biodiversity attributes, e.g. related to species productivity, seasonality and phenology (Pettorelli et al., 2016; Vaz et al., 2018b). Our study is a step-forward for advancing multidisciplinary approaches towards nature's benefits to people, constituting a great opportunity to inform decision-makers and managers on priority areas for the timely management of non-native trees from a cultural perspective.

6.5. Conclusions

We proposed a pioneer approach that combines remote sensing data from social media and Earth observations in a multimodel framework to assess the spatiotemporal contributions of non-native trees to cultural ecosystem services. The approach was applied in a National Park of Portugal, where non-native trees were found to contribute more to cultural benefits (than native trees) in accessible areas with colourful and homogeneous landscapes, during Autumn and Spring. In Winter, native trees showed higher contributions, particularly in remote areas. In Summer, non-native and native trees showed similar contributions. Our results are congruent with the phenology of dominant vegetation and match differences in the seasonality of tourist demand. The proposed approach is replicable worldwide, supporting decision-making on biosecurity efforts to safeguard the natural and cultural heritage in the Anthropocene.

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SUPPLEMENTARY MATERIAL V

Appendix A - The calculation of cover areas of non-native and native tree species

The calculation of the areas covered by non-native and native tree species in each 1 km^2 grid cell from the test area, was grounded on data from COS2007 (available at: http://www.dgterritorio.pt/cartografia_e_geodesia/cartografia/cartografia_tematica/carta_de_ocupacao_do_solo__cos_/cos__2007/). This source includes the most recent and available land cover data provided as shapefile for the country, in hectares (ha). Since the land cover data is not provided at the species level, we considered a conservative approach in the sense that only classes dominated by non-native or native land cover were considered (following (Vaz et al., 2018; see Table S6.1). To assess the area covered by non-native and native trees per grid cell, the information from COS2007 was merged with a polygon grid created for the test area, in ArcMap 10.3 (ESRI, 2014). The final values for the test area were compared to the values provided by the Portuguese National Forest Inventory (ICNF, 2013), and the obtained proportions of non-native and native tree cover areas were validated.

Table S6.1. Description of the land cover levels available in COS 2007 with indication of the native (NT) and non-native (NNT) type of land-cover.

Level 3	Level 4	Level 5	Description	NT/NNT
Agro-forestry systems (2.4.4)				
2.4.4	2.4.4.01	2.4.4.01.1	*Quercus suber* w/ dry cultures	NT
2.4.4	2.4.4.01	2.4.4.01.2	*Q. ilex* w/ dry cultures	NT
2.4.4	2.4.4.01	2.4.4.01.3	Other *Quercus* w/ dry cultures	NT
2.4.4	2.4.4.01	2.4.4.01.5	*Q. suber* and *Q. ilex* w/ dry cultures	NT
2.4.4	2.4.4.02	2.4.4.02.1	*Q. suber* w/ irrigated cultures	NT
2.4.4	2.4.4.02	2.4.4.02.2	*Q. ilex* w/ irrigated cultures	NT
2.4.4	2.4.4.02	2.4.4.02.3	Other *Quercus* w/ irrigated cultures	NT
2.4.4	2.4.4.02	2.4.4.02.5	*Q. suber* and *Q. ilex* w/ irrigated cultures	NT
2.4.4	2.4.4.03	2.4.4.03.1	*Q. suber* w/ pastures	NT
2.4.4	2.4.4.03	2.4.4.03.2	*Q. ilex* w/ pastures	NT
2.4.4	2.4.4.03	2.4.4.03.3	Other *Quercus* w/ pastures	NT
2.4.4	2.4.4.03	2.4.4.03.5	*Q. suber* and *Q. ilex* w/ pastures	NT
2.4.4	2.4.4.04	2.4.4.04.1	*Q. suber* w/ permanent cultures	NT
2.4.4	2.4.4.04	2.4.4.04.2	*Q. ilex* w/ permanent cultures	NT
2.4.4	2.4.4.04	2.4.4.04.3	Other *Quercus* w/ permanent cultures	NT
2.4.4	2.4.4.04	2.4.4.04.5	*Q. suber* and *Q. ilex* w/ permanent cultures	NT
Broadleaved forests (3.1.1)				

3.1.1	3.1.1.01	3.1.1.01.1	Pure *Q. suber* forests	NT
3.1.1	3.1.1.01	3.1.1.01.2	Pure *Q. ilex* forests	NT
3.1.1	3.1.1.01	3.1.1.01.3	Pure Other *Quercus* forests	NT
3.1.1	3.1.1.01	3.1.1.01.5	Pure *Eucalyptus* forests	NNT
3.1.1	3.1.1.01	3.1.1.01.6	Pure Invasive species forests	NNT
3.1.1	3.1.1.02	3.1.1.02.1	Dominated *Q. suber* forests	NT
3.1.1	3.1.1.02	3.1.1.02.2	Dominated *Q. ilex* forests	NT
3.1.1	3.1.1.02	3.1.1.02.3	Dominated Other *Quercus* forests	NT
3.1.1	3.1.1.02	3.1.1.02.5	Dominated *Eucalyptus* forests	NNT
3.1.1	3.1.1.02	3.1.1.02.6	Dominated Invasive species forests	NNT
Coniferous forests (3.1.2)				
3.1.2	3.1.2.01	3.1.2.01.1	Pure *Pinus pinaster* forests	NNT
3.1.2	3.1.2.01	3.1.2.01.2	Pure *P. pinea* forests	NNT
3.1.2	3.1.2.02	3.1.2.02.1	Dominated *P. pinaster* forests	NNT
3.1.2	3.1.2.02	3.1.2.02.2	Dominated *P. pinea* forests	NNT
Mixed forests (broadleaved w/ coniferous) (3.1.3)				
3.1.3	3.1.3.01	3.1.3.01.1	Mixed *Q. suber* w/ coniferous	NT/NNT
3.1.3	3.1.3.01	3.1.3.01.2	Mixed *Q. ilex* w/ coniferous	NT/NNT
3.1.3	3.1.3.01	3.1.3.01.3	Mixed *Quercus* w/ coniferous	NT/NNT
3.1.3	3.1.3.01	3.1.3.01.5	Mixed *Eucalyptus* w/ coniferous	NNT
3.1.3	3.1.3.01	3.1.3.01.6	Mixed Invasive species w/ coniferous	NNT
Open forests (3.2.4)				
3.2.4	3.2.4.01	3.2.4.01.1	Open *Q. suber* forests	NT
3.2.4	3.2.4.01	3.2.4.01.2	Open *Q. ilex* forests	NT
3.2.4	3.2.4.01	3.2.4.01.3	Open Other *Quercus* forests	NT
3.2.4	3.2.4.01	3.2.4.01.5	Open *Eucalyptus* forests	NNT
3.2.4	3.2.4.01	3.2.4.01.6	Open Invasive species forests	NNT
3.2.4	3.2.4.02	3.2.4.02.1	Open dominated *Q. suber* forests	NT
3.2.4	3.2.4.02	3.2.4.02.2	Open dominated *Q. ilex* forests	NT
3.2.4	3.2.4.02	3.2.4.02.3	Open dominated Other *Quercus* forests	NT
3.2.4	3.2.4.02	3.2.4.02.5	Open dominated *Eucalyptus* forests	NNT
3.2.4	3.2.4.02	3.2.4.02.6	Open dominated Invasive species forests	NNT
3.2.4	3.2.4.03	3.2.4.03.1	Open *P. pinaster* forests	NNT
3.2.4	3.2.4.03	3.2.4.03.2	Open *P. pinea* forests	NNT
3.2.4	3.2.4.04	3.2.4.04.1	Mixed *Pinus pinaster* forests w/ coniferous	NNT
3.2.4	3.2.4.04	3.2.4.04.2	Mixed *P. pinea* forests w/ coniferous	NNT
3.2.4	3.2.4.05	3.2.4.05.1	Open *Q. suber* forests w/ coniferous	NT
3.2.4	3.2.4.05	3.2.4.05.2	Open *Q. ilex* forests w/ coniferous	NT
3.2.4	3.2.4.05	3.2.4.05.3	Open Other *Quercus* forests w/ coniferous	NT
3.2.4	3.2.4.05	3.2.4.05.5	Open *Eucalyptus* forests w/ coniferous	NNT
3.2.4	3.2.4.05	3.2.4.05.6	Open Invasive species forests w/ coniferous	NNT

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Appendix B - Mining data from social media

Cultural services were evaluated through the screening of photographs from two popular social media platforms: Flickr and Wikiloc. Flickr (at: https://www.flickr.com) is trendy among photographers to capture aesthetic and inspirational values of ecosystems. Wikiloc (at: http://www.wikiloc.com) shows nature routes illustrated by photographs related to touristic and recreational activities in the wild (e.g. hiking, cycling, walking).

Many social media platforms provide an application programming interface (API) that can be used to collect publicly available information published by the users. In this research, we collected geographically referenced social media data from Flickr that were posted prior to the year 2018, within the borders of our study area. Specifically, we used the Flickr API (https://www.flickr.com/services/api/), specifying a time window and a bounding box with pair of coordinates (in our case: minimum latitude: 41.653104; maximum lat.: 42.083595; min. longitude: -8.426270; max. lon.: -7.754076) around the test area. This information was stored as an excel file with the following attributes: user-id, date taken, latitude, longitude, picture uniform source locator (url). Social media data collection tools were developed by Paulo Pereira (pauloa.d.pereira@gmail.com) for Python 3.5. The code and details can be freely accessed here: https://github.com/PJADPereira/flickr-download (accessed 07.05.2018).

For Wikiloc, however, we used Google Earth extension to manually check all tracks and photographs uploaded by the users. This was done by delimitating our manual search to the boundaries of the test area, uploaded as a kmz file (Figure S6.1).

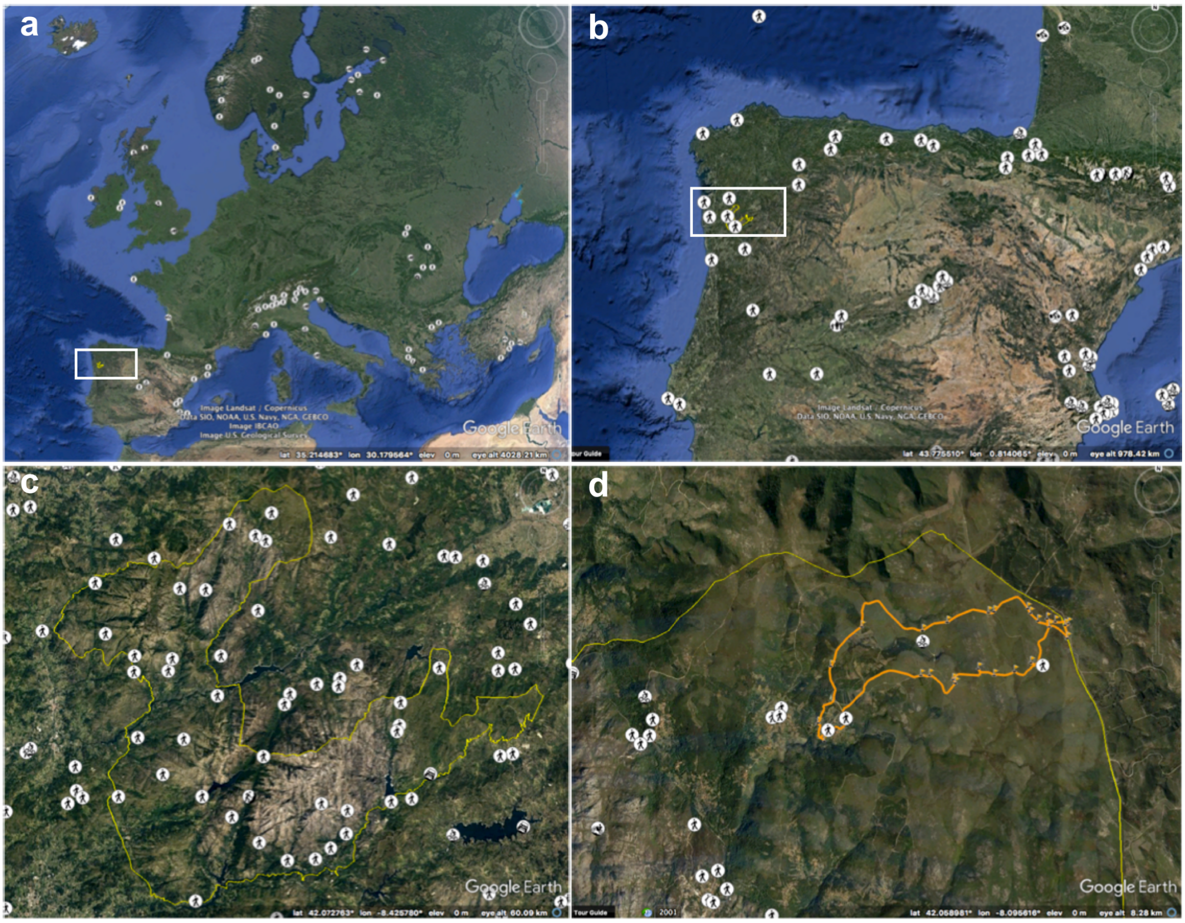

Figure S6.1. Overview of the location of the test area in Google Earth, at the European (a) and Iberian (b) context. The figure also shows the location of some Wikiloc tracks in the test area (c); and one Wikiloc track as example (d), in which the yellow flags represent georeferenced places with photographs.

Appendix C - Information on predictors

We show information on the initial set of predictors considered in our research. The code and description of all predictors is shown in Table S6.2.

Accessibility variables

Besides the static variables revealed in Table 6.2 (main text) we also assessed the aspect of each grid cell (North, South, West, East) and the number of fires, as well as the distance of each grid cell to the rivers and roads, which were not considered in the main study due to high correlation values (> 0.60) with other variables. Details on the static predictors can be found in (Vicente et al., 2016)

Seasonal Ecosystem functioning variables

The MODerate Resolution Imaging Spectroradiometer (MODIS) MOD13Q1 product provides 16-day composite vegetation indices (VI) at 250 m spatial resolution, making available, consistent, spatial and temporal time series of global vegetation conditions suitable for characterizing the annual seasonal dynamics of ecosystems and its spatial heterogeneity (Alcaraz et al., 2006; Cabello et al., 2012; Paruelo et al., 2001). The Enhanced Vegetation Index (EVI), available from MOD13Q1 product (version 6), was used in this study since it provides improved sensitivity over high-biomass regions and greater vegetation monitoring ability through the decoupling of the canopy background signal and a reduction in atmosphere influences (Didan et al., 2015; Huete et al., 2002). The EVI time-series was comprised from December 2011 to February 2017 to coincide with the years for which higher availability of pictures occur. Data was downloaded from the EarthData platform (URL: https://search.earthdata.nasa.gov/) and then mosaicked and re-projected to WGS 1984 - UTM 29N coordinate system using MODIS Reprojection Tool (MRT release 4.1, 2011, URL: https://lpdaac.usgs.gov/tools/modis_reprojection_tool).

In order to improve the retrieval of meaningful information from the EVI time series, to remove outliers or spurious values, and to increase the signal-to-noise ratio, two procedures were performed sequentially: (1) Hampel outlier filtering (Davies and Gather, 1993; Hampel, 1974) and, (2) time series smoothing using the Whittaker-Henderson algorithm (attributed to Whittaker, 1923) with upper envelope weighting.

From this pre-processed series, we computed two measures to characterize ecosystem dynamics in each meteorological season based on the statistical properties of annual VI curves. These measures were the seasonal average (allowing to characterize vegetation amount) and the seasonal amplitude (describing vegetation seasonal change). To depict the spatial heterogeneity at the landscape level we used the standard-deviation and the mean to aggregate values from MODIS original spatial resolution from 250 m to 1000 m (grid cell size). All analytical procedures were performed in R software v3.4.0 (R Development Core Team, 2017).

Spectral measures of landscape heterogeneity

In order to describe seasonal landscape heterogeneity and diversity, with a special emphasis on the visible part of the electromagnetic spectrum (which we assume to be closely related to human vision and perception), we used Sentinel-2 L1C images available in Google Earth Engine (GEE; Gorelick et al., 2017). In the GEE platform, we started by developing multi-annual cloud-free seasonal composites (one for each meteorological season) by applying a median reducer to all scenes within each season with less than 30 % of cloud cover and between 2015 and 2018 (containing all available data in the Sentinel-2 archive). Using these data, and in order to portray spectral heterogeneity, we calculated the standard-deviation and the average of reflectance values for bands 2 (blue), 3 (green) and 4 (red) aggregating pixel values from the original spatial resolution at 10 m to 1000 m (final grid cell size).

From each composite and to additionally characterize land surface heterogeneity and diversity, we performed a k-means unsupervised classification using Sentinel-2 bands 2, 3 and 4. We set the algorithm to obtain a total of 20 clusters. Based on the k-means classified data for each seasonal composite, we calculated the following diversity metrics for the 1000 m units: (1) number of clusters, (2) Shannon diversity index, (3) the reciprocal Simpson diversity index, and (4) the inverse Simpson diversity index.

Table S6.2. List of initial predictors considered (acronyms and descriptions).

	Acronym	Description
Accessibility and remoteness	Effort	Effort index
	RoadDist	Distance to main roads
	RivDist	Distance to main rivers
	Altitude	Altitude
	Slope08	Slope from 0 to 8 degrees
	Slope815	Slope from 8-15 degrees
	Slope1525	Slope from 15-25 degrees
	Slope25	Slope from higher than 25 degrees
	AspectFlat	Aspect
	AspectNort	Northern aspect
	AspectEste	Eastern aspect
	AspectSul	Southern aspect
	AspectOest	Western aspect
	ViewshedNu	Viewshed dimension (as proxy for visual accessibility) based on the number of 30m pixels from the centroid the area (based on altimetry from GDEM)
	NumFires97	Number of fires until 1990-1997
	NumFires_1	Number of fires after 1997
	NumFires90	Number of fires before 1990
	RoadDens	Density of all roads
	RivDens	Density of all rivers
Landscape functioning	EVI_WINT_tsMN_mean	Mean annual EVI (enhanced vegetation index) in the Winter (based on 2011-2017 Modis time series)
	EVI_SPRG_tsMN_mean	Mean annual EVI (enhanced vegetation index) in the Spring (based on 2011-2017 Modis time series)
	EVI_SUMM_tsMN_mean	Mean annual EVI (enhanced vegetation index) in the Summer (based on 2011-2017 Modis time series)
	EVI_AUTM_tsMN_mean	Mean annual EVI (enhanced vegetation index) in the Autumn (based on 2011-2017 Modis time series)
	EVI_WINT_tsMN_sd	Seasonal variability of EVI (enhanced vegetation index) in the Winter (based on 2011-2017 Modis time series)
	EVI_SPRG_tsMN_sd	Seasonal variability of EVI (enhanced vegetation index) in the Spring (based on 2011-2017 Modis time series)
	EVI_SUMM_tsMN_sd	Seasonal variability of EVI (enhanced vegetation index) in the Summer (based on 2011-2017 Modis time series)
	EVI_AUTM_tsMN_sd	Seasonal variability of EVI (enhanced vegetation index) in the Autumn (based on 2011-2017 Modis time series)

	EVI_WINT_tsRG_mean	Mean spatial EVI (enhanced vegetation index) in the Winter (based on 2011-2017 Modis time series)
	EVI_SPRG_tsRG_mean	Mean spatial EVI (enhanced vegetation index) in the Spring (based on 2011-2017 Modis time series)
	EVI_SUMM_tsRG_mean	Mean spatial EVI (enhanced vegetation index) in the Summer (based on 2011-2017 Modis time series)
	EVI_AUTM_tsRG_mean	Mean spatial EVI (enhanced vegetation index) in the Autumn (based on 2011-2017 Modis time series)
	EVI_WINT_tsRG_sd	Spatial heterogeneity of EVI (enhanced vegetation index) in the Winter (based on 2011-2017 Modis time series)
	EVI_SPRG_tsRG_sd	Spatial heterogeneity of EVI (enhanced vegetation index) in the Spring (based on 2011-2017 Modis time series)
	EVI_SUMM_tsRG_sd	Spatial heterogeneity of EVI (enhanced vegetation index) in the Summer (based on 2011-2017 Modis time series)
	EVI_AUTM_tsRG_sd	Spatial heterogeneity of EVI (enhanced vegetation index) in the Autumn (based on 2011-2017 Modis time series)
Landscape spatial heterogeneity	S2_autm_clust.1_nclust	Number of clusters (k-mean classification) in the area during Autumn (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_sprg_clust.1_nclust	Number of clusters (k-mean classification) in the area during Spring (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_summ_clust.1_nclust	Number of clusters (k-mean classification) in the area during Summer (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_wint_clust.1_nclust	Number of clusters (k-mean classification) in the area during Winter (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_autm_clust.1_SHDI	Shannon-diversity (k-mean classification) in the area during Autumn (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_sprg_clust.1_SHDI	Shannon-diversity (k-mean classification) in the area during Spring (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_summ_clust.1_SHDI	Shannon-diversity (k-mean classification) in the area during Summer (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_wint_clust.1_SHDI	Shannon-diversity (k-mean classification) in the area during Winter (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_autm_clust.1_cSDI	Simpson-diversity (k-mean classification) in the area during Autumn (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_sprg_clust.1_cSDI	Simpson-diversity (k-mean classification) in the area during Spring (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_summ_clust.1_cSDI	Simpson-diversity (k-mean classification) in the area during Summer (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_wint_clust.1_cSDI	Simpson-diversity (k-mean classification) in the area during Winter (based on the multi-annual, cloud-free mosaics from Sentinel-2)

	S2_autm_clust.1_iSDI	Inverse of Simpson-diversity (k-mean classification) in the area during Autumn (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_sprg_clust.1_iSDI	Inverse of Simpson-diversity (k-mean classification) in the area during Spring (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_summ_clust.1_iSDI	Inverse of Simpson-diversity (k-mean classification) in the area during Summer (based on the multi-annual, cloud-free mosaics from Sentinel-2)
	S2_wint_clust.1_iSDI	Inverse of Simpson-diversity (k-mean classification) in the area during Winter (based on the multi-annual, cloud-free mosaics from Sentinel-2)
Landscape color diversity	S2_autm_mosaic.2_mean	Mean spectral diversity (B; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Autumn
	S2_autm_mosaic.3_mean	Mean spectral diversity (G; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Autumn
	S2_autm_mosaic.4_mean	Mean spectral diversity (R; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Autumn
	S2_autm_mosaic.2_sd	Heterogeneity of spectral diversity (B; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Autumn
	S2_autm_mosaic.3_sd	Heterogeneity of spectral diversity (G; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Autumn
	S2_autm_mosaic.4_sd	Heterogeneity of spectral diversity (R; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Autumn
	S2_sprg_mosaic.2_mean	Mean spectral diversity (B; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Spring
	S2_sprg_mosaic.3_mean	Mean spectral diversity (G; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Spring
	S2_sprg_mosaic.4_mean	Mean spectral diversity (R; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Spring
	S2_sprg_mosaic.2_sd	Heterogeneity of spectral diversity (B; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Spring
	S2_sprg_mosaic.3_sd	Heterogeneity of spectral diversity (G; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Spring
	S2_sprg_mosaic.4_sd	Heterogeneity of spectral diversity (R; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Spring
	S2_summ_mosaic.2_mean	Mean spectral diversity (B; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Summer
	S2_summ_mosaic.3_mean	Mean spectral diversity (G; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Summer
	S2_summ_mosaic.4_mean	Mean spectral diversity (R; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Summer
	S2_summ_mosaic.2_sd	Heterogeneity of spectral diversity (B; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Summer

S2_summ_mosaic.3_sd	Heterogeneity of spectral diversity (G; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Summer
S2_summ_mosaic.4_sd	Heterogeneity of spectral diversity (R; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Summer
S2_wint_mosaic.2_mean	Mean spectral diversity (B; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Winter
S2_wint_mosaic.3_mean	Mean spectral diversity (G; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Winter
S2_wint_mosaic.4_mean	Mean spectral diversity (R; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Winter
S2_wint_mosaic.2_sd	Heterogeneity of spectral diversity (B; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Winter
S2_wint_mosaic.3_sd	Heterogeneity of spectral diversity (G; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Winter
S2_wint_mosaic.4_sd	Heterogeneity of spectral diversity (R; Sentinel-2 mosaics from multi-annual, cloud-free imagery) for Winter

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Appendix D - Details on the calculation of the odds ratio

We followed the same procedure from Vaz et al. (2018) and used the odds ratio as an indicator of the contribution of non-native tree species on cultural ecosystem services. The odds ratio is an effect size statistic, often applied in meta-analysis and case-control studies, as a measure of association between an exposure and an outcome, against the frequency of such outcome if expected by chance (Borenstein et al., 2008). In our case, the odds ratio expresses the direction of contributions from of non-native tree species (i.e., exposure) in each photograph of the cultural service (i.e., outcome), compared to the effect of native trees (i.e., non-exposure or comparator). The computation of the direction of contributions was further achieved considering the frequency of non-native and native trees in a data source against their prevalence, as cover area, in each grid cell (i.e., expected by change), as the control situation (see Appendix A). For computing the odds ratio, we first organised the information of each grid cell in contingency tables (Table S6.3).

Table S6.3. Example of a contingency table for calculating the OR, based on the collection of information of non-native and native tree, for cultural ecosystem services.

		Exposure	
		Exposure (non-native trees)	Non-exposure (native trees)
Outcome	Proportion of photographs dominated by non-native and native trees	*A*	*B*
	Proportion of area covered by non-native and native trees	*C*	*D*

For the calculation of the odds ratio in each cell, we used Peto's method, which is grounded on the following statistical procedures (Borenstein et al., 2008; Viechtbauer, 2010):

$$\Psi = \exp(O\text{-}E/V)$$

$$O = A$$

$$E = (A+B) / (A+C)/n$$

$$V = (A+B)(C+D)(A+C)(B+D) / n^2(n-1)$$

Where Ψ is Peto's odds ratio, n = A+B+C+D, and V is both weighting factor and variance for the difference between observed and expected A, O-E.

The odds ratio was log-transformed (logOR), so that positive and negative values of logOR indicate that the contribution of non-native trees to cultural services, in comparison to the contribution of native trees, is respectively higher or lower than their proportion in the analysed grid cell. Thus, logOR values higher or lower than 0 respectively indicate a positive or negative significant preference for non-native trees on a cultural service. LogOR equal to 0 indicate non-significant effects of non-native trees, i.e. that both non-native and native trees were similarly frequent in the photoseries.

The expected-by-chance data was calculated considering the area covered by non-native and native trees in each grid cell under analysis. Since none of the two reference values (C and D; Table S6.3) can be zero, if a given grid cell showed a value = 0, or whether non-native or native trees were absent, the odds ratio for that grid cell was not computed. Also, land cover data was in different orders of magnitude compared to the number of photographs, which could have resulted in unbalanced contingency tables. This means that the sum of values in a row of the contingency table (A and B; Table S6.3) and the sum of values in the other row (C and D; Table S6.3) differed in their magnitude orders. Since Peto's method may fail in unbalanced contingency tables (Sweeting et al., 2004), we re-calculated the values used to calculate the expected ratio (C and D) dividing them by a constant that makes their sum equal to A + B while keeping the original ratio between non-native and native tree cover areas.

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Appendix E - Proportion of tree species in the photoseries

Table S6.4. Number and proportion of photographs dominated or co-dominated by non-native tree species across the set of 1748 analysed photographs.

Genus	Number	Proportion
Pinus pinaster	602	53.23
Pseudotsuga menziesii	96	8.49
Chamaecyparis lawsoniana	92	8.13
Acacia dealbata	85	7.52
Acacia longifolia	79	6.98
Eucalyptus globulus	49	4.33
Cupressus lusitanica	41	3.63
Other-Ornamental	34	3.01
Sequoiadendron spp.	11	0.97
Quercus coccinea	9	0.80
Citrus spp.	6	0.53
Tilia cordata	5	0.44
Acer japonicum	4	0.35
Arecaceae spp.	3	0.27
Platanus spp.	3	0.27
Betula pendula	2	0.18
Abies spp.	1	0.09
Ailanthus altissima	4	0.35
Camelia sinensis	1	0.09
Ginkgo biloba	1	0.09
Magnolia spp.	1	0.09
Robinia pseudoacacia	1	0.09
Salix babylonica	1	0.09

Appendix F - Results of the multidimensional unfolding

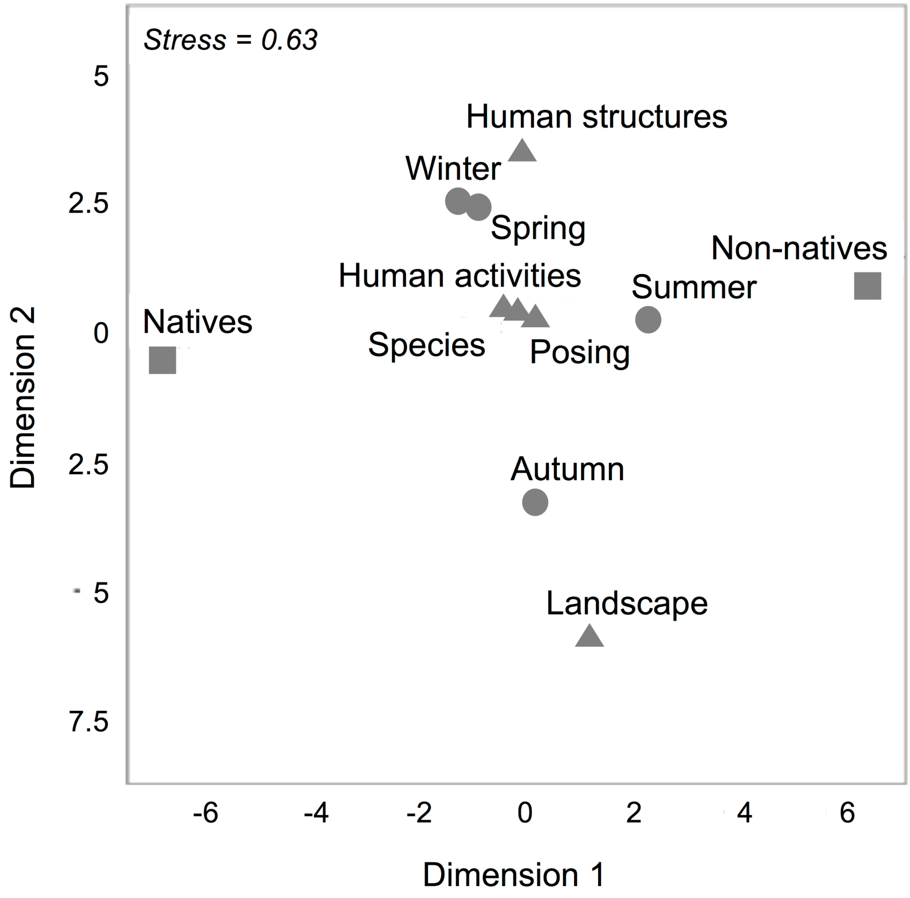

Figure S6.2. Representation of results of the multidimensional unfolding (MDU; (Mair et al., 2015); stress = 0.63) across the ordination axes 1 and 2, considering the number of photographs dominated by non-native or native trees (squares), and the classification of photographs in agreement to the season (circles) and focus of the photograph (triangles). The MDU was used to visualise the association between the date, focus, and type of dominant tree in each photograph, using R software (Elff, 2016).

REFERENCE

Elff, M., 2016. Metric Unfolding. R package.

Appendix G - Details on the results from the computation of odds ratio across grid cells

For each grid cell, we computed the log odds ratio (logOR; Appendix D) under Peto's method (see Figure S6.3). We then computed the weighted logOR under the DerSimonian-Laird random effects model, with corresponding lower (Lower CI) and upper (Upper CI) 95% confidence intervals (Table S6.5). The significance of logORw was obtained though non-parametric permutation tests (under 1000 iteractions). The significance of the heterogeneity of computed logOR for each data source is also shown and was obtained from a chi-squared test of the Q-statistic (QT). Computations were done considering all data, and each meteorological season indvidually.

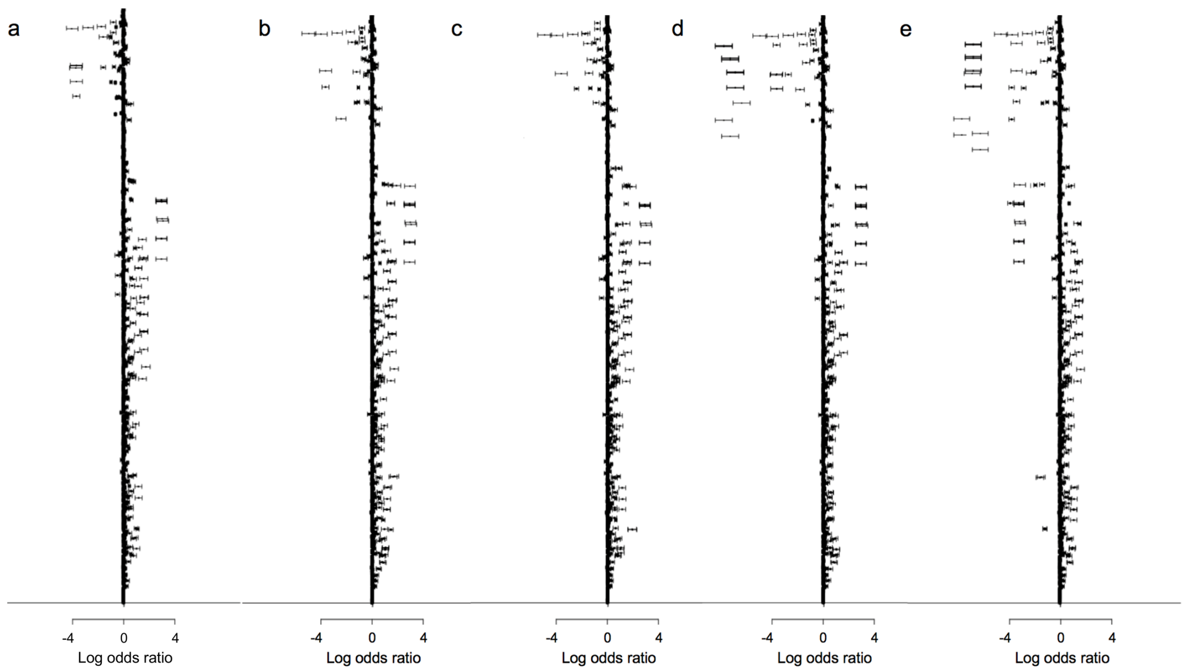

Figure S6.3. The distribution of the logarithmic odds ratio across the set of grid cells from the test area, considering all data (a), and each meteorological season of the year: autumn (b), spring (c), summer (d) and winter (e). The bars around each odds ratio, represent the lower and upper confidence intervals (ranging from -4 to 4).

Table S6.5. Results of the calculation of the weighted log odds ratio (wlogOR) for all data and each meteorological season (n=1008). Values higher or lower than zero respectively suggest that non-native trees contribute more or less to cultural service, in contrast to native trees. St. error is the standard error, Degree is the number of degrees of freedom. QT values indicate the heterogeneity of the log odds ratio across the grid cells of the study area, tested by means of the Q statistics. The lower and upper confidence intervals (CI) are indicated.

	wlogOR	**St. error**	**z-value**	**p-value**	**Lower CI**	**Upper CI**	**Degree**	**QT**
All data	0.584	0.059	9.819	0.001	0.467	0.700	675	428.304
Autumn	0.227	0.072	3.182	0.002	0.087	0.367	675	119.475
Spring	0.187	0.073	2.572	0.001	0.044	0.329	675	105.621
Summer	0.244	0.072	3.405	0.100	0.104	0.385	675	109.750
Winter	-0.197	0.072	2.727	0.001	-0.055	-0.339	675	124.665

Appendix H - Details on results of competing models and respective predictors

We considered three competing models (M1, M2, M3), grounded on a single group or combinations of both groups of predictors (i.e. environmental context versus landscape setting): M1 - predictors expressing the environmental context of each grid cell in terms of accessibility and wilderness; M2 - predictors expressing landscape features related to spatial diversity, colour heterogeneity or functioning; or, M3 - a combination of both sets of predictors.

We tested the predictive power of each model using multimodel inference (MMI; (Burnham and Anderson, 2003) and averaging procedures based on Akaike Information Criteria differences (ΔAICc; Burnham et al., 2011). We implemented random-effect meta-regression models, using the maximum likelihood estimation, in R software (glmulti package; Calcagno, 2013). Models with differences between corrected AIC (ΔAICc) < 2 were considered as having highest explanatory power and support, in comparison to the remaining models (Anderson, 2007). The multimodel procedure was applied considering each meteorological season individually.

Below we show the combination of predictors within each competing model, with AICc < 2, as well as the general contribution of each predictor from the best competing model to explain the variation in the contribution of non-native trees to cultural services (Figure S6.4-S6.7). We also show the full results of the multimodel framework below, in Table S6.6.

Table S6.6. Combination of predictors within each competing model, with ΔAICc < 2, for each meteorological season.

Predictor combinations	**AICc**
Autumn - M1	
yi ~ 1 + RivDens + Altitude	2122.780
yi ~ 1 + RivDens	2123.350
yi ~ 1 + Altitude	2123.625
yi ~ 1 + RoadDens + RivDens	2123.642
yi ~ 1 + RivDens + AspectOest	2123.713
yi ~ 1 + Altitude + AspectOest	2124.219
yi ~ 1 + RivDens + ViewshedNu	2124.318
yi ~ 1 + RivDens + AspectNort	2124.473
yi ~ 1 + RivDens + Altitude + AspectOest	2124.479
yi ~ 1 + RoadDens	2124.492
yi ~ 1 + RoadDens + Altitude	2124.587
yi ~ 1 + Altitude + ViewshedNu	2124.618

yi ~ 1 + RoadDens + RivDens + AspectOest	2124.633
yi ~ 1 + RivDens + NumFires_1	2124.703
yi ~ 1 + RivDens + Slope815	2124.728
Autumn - M2	
yi ~ 1 + S2_autm_clust1_nclust + S2_autm_mosaic3_sd	2123.813
yi ~ 1 + S2_autm_mosaic3_sd	2124.332
yi ~ 1 + S2_autm_clust1_nclust	2124.711
yi ~ 1 + S2_autm_clust1_SHDI + S2_autm_mosaic3_sd	2124.999
yi ~ 1 + EVI_AUTM_tsRG_mean + S2_autm_mosaic3_sd	2125.363
yi ~ 1 + S2_autm_clust1_nclust + S2_autm_clust1_SHDI	2125.487
yi ~ 1 + EVI_AUTM_tsMN_sd + S2_autm_mosaic3_sd	2125.515
yi ~ 1 + EVI_AUTM_tsMN_mean + S2_autm_mosaic3_sd	2125.593
yi ~ 1 + S2_autm_clust1_SHDI	2125.614
yi ~ 1 + EVI_AUTM_tsRG_mean + S2_autm_clust1_nclust	2125.682
yi ~ 1 + S2_autm_mosaic2_mean + S2_autm_mosaic3_sd	2125.722
yi ~ 1 + S2_autm_mosaic2_sd + S2_autm_mosaic3_sd	2125.791
yi ~ 1 + S2_autm_mosaic3_sd + S2_autm_mosaic4_sd	2125.800
yi ~ 1 + EVI_AUTM_tsRG_sd + S2_autm_mosaic3_sd	2125.804
Autumn - M3	
yi ~ 1 + RivDens + S2_autm_mosaic3_sd	2122.711
yi ~ 1 + RivDens	2122.780
yi ~ 1 + Altitude	2123.350
yi ~ 1 + RivDens + AspectOest + S2_autm_mosaic3_sd	2123.439
yi ~ 1 + Altitude + S2_autm_mosaic3_sd	2123.528
yi ~ 1 + RivDens + S2_autm_clust1_nclust	2123.540
yi ~ 1 + RivDens + Altitude	2123.625
yi ~ 1 + RoadDens + RivDens	2123.642
yi ~ 1 + RivDens + AspectOest	2123.713
yi ~ 1 + S2_autm_mosaic3_sd	2123.813
yi ~ 1 + RoadDens + RivDens + S2_autm_mosaic3_sd	2123.821
yi ~ 1 + RivDens + S2_autm_clust1_SHDI	2123.900
yi ~ 1 + RivDens + S2_autm_clust1_SHDI + S2_autm_mosaic3_sd	2123.903
yi ~ 1 + RivDens + Altitude + S2_autm_mosaic3_sd	2123.952
yi ~ 1 + Altitude + S2_autm_clust1_nclust	2124.014
yi ~ 1 + RivDens + S2_autm_clust1_nclust + S2_autm_mosaic3_sd	2124.155
yi ~ 1 + Altitude + AspectOest + S2_autm_mosaic3_sd	2124.213
yi ~ 1 + Altitude + AspectOest	2124.219
yi ~ 1 + RivDens + AspectOest + S2_autm_clust1_nclust	2124.228
yi ~ 1 + RoadDens + S2_autm_mosaic3_sd	2124.240
yi ~ 1 + RivDens + ViewshedNu	2124.318
yi ~ 1 + S2_autm_clust1_nclust	2124.332
yi ~ 1 + RivDens + ViewshedNu + S2_autm_mosaic3_sd	2124.385

yi ~ 1 + RivDens + EVI_AUTM_tsRG_mean + S2_autm_mosaic3_sd	2124.398
yi ~ 1 + RivDens + EVI_AUTM_tsMN_sd + S2_autm_mosaic3_sd	2124.436
yi ~ 1 + RivDens + EVI_AUTM_tsRG_mean	2124.466
yi ~ 1 + RivDens + AspectNort + S2_autm_mosaic3_sd	2124.472
yi ~ 1 + RivDens + AspectNort	2124.473
yi ~ 1 + RivDens + Altitude + AspectOest	2124.479
yi ~ 1 + RivDens + Slope815 + S2_autm_mosaic3_sd	2124.487
yi ~ 1 + RoadDens	2124.492
yi ~ 1 + RivDens + AspectOest + S2_autm_clust1_SHDI + S2_autm_mosaic3_sd	2124.514
yi ~ 1 + RivDens + NumFires_1 + S2_autm_mosaic3_sd	2124.534
yi ~ 1 + RivDens + EVI_AUTM_tsMN_sd	2124.555
yi ~ 1 + RoadDens + Altitude	2124.587
yi ~ 1 + Altitude + S2_autm_clust1_SHDI	2124.593
yi ~ 1 + RivDens + EVI_AUTM_tsMN_mean + S2_autm_mosaic3_sd	2124.593
yi ~ 1 + AspectOest + S2_autm_mosaic3_sd	2124.604
yi ~ 1 + RivDens + S2_autm_mosaic2_mean + S2_autm_mosaic3_sd	2124.610
yi ~ 1 + RoadDens + RivDens + AspectOest + S2_autm_mosaic3_sd	2124.616
yi ~ 1 + Altitude + ViewshedNu	2124.618
yi ~ 1 + RivDens + Altitude + AspectOest + S2_autm_mosaic3_sd	2124.629
yi ~ 1 + Altitude + AspectOest + S2_autm_clust1_nclust	2124.631
yi ~ 1 + RoadDens + RivDens + AspectOest	2124.633
yi ~ 1 + RivDens + Altitude + S2_autm_clust1_nclust	2124.656
yi ~ 1 + RivDens + EVI_AUTM_tsMN_mean	2124.670
yi ~ 1 + RivDens + S2_autm_mosaic3_sd + S2_autm_mosaic4_sd	2124.690
yi ~ 1 + RivDens + S2_autm_mosaic2_mean	2124.698
yi ~ 1 + RivDens + S2_autm_clust1_nclust + S2_autm_clust1_SHDI	2124.699
yi ~ 1 + RivDens + NumFires_1	2124.703
yi ~ 1 + S2_autm_clust1_nclust + S2_autm_mosaic3_sd	2124.711
Spring - M1	
yi ~ 1 + effort + RoadDens	2124.616
yi ~ 1 + RoadDens	2124.870
yi ~ 1 + effort	2124.972
yi ~ 1 + RivDens	2125.494
yi ~ 1 + AspectNort	2125.674
yi ~ 1 + Altitude	2125.779
yi ~ 1 + effort + RivDens	2125.879
yi ~ 1 + NumFires_1	2126.040
yi ~ 1 + effort + RoadDens + RivDens	2126.080
yi ~ 1 + effort + Altitude	2126.109
yi ~ 1 + ViewshedNu	2126.122
Spring - M2	
yi ~ 1 + S2_sprg_clust1_SHDI + S2_sprg_mosaic2_mean	2123.454

yi ~ 1 + S2_sprg_clust1_SHDI	2124.535
yi ~ 1 + EVI_SPRG_tsRG_sd + S2_sprg_clust1_SHDI	2124.695
yi ~ 1 + S2_sprg_clust1_nclust + S2_sprg_clust1_SHDI	2124.747
yi ~ 1 + S2_sprg_clust1_SHDI + S2_sprg_mosaic3_sd	2124.820
yi ~ 1 + S2_sprg_clust1_SHDI + S2_sprg_mosaic4_sd	2125.084
yi ~ 1 + EVI_SPRG_tsMN_sd + S2_sprg_clust1_SHDI	2125.122
yi ~ 1 + EVI_SPRG_tsRG_mean + S2_sprg_clust1_SHDI	2125.159
yi ~ 1 + S2_sprg_clust1_SHDI + S2_sprg_mosaic2_sd	2125.228
yi ~ 1 + S2_sprg_mosaic2_mean	2125.236
yi ~ 1 + EVI_SPRG_tsRG_sd	2125.307
Spring - M3	
yi ~ 1 + RoadDens + S2_sprg_clust1_SHDI	2123.354
yi ~ 1 + S2_sprg_clust1_SHDI	2124.085
yi ~ 1 + effort + S2_sprg_clust1_SHDI	2124.495
yi ~ 1 + S2_sprg_clust1_SHDI + S2_sprg_mosaic2_mean	2124.535
yi ~ 1 + RoadDens	2124.616
yi ~ 1 + EVI_SPRG_tsRG_sd + S2_sprg_clust1_SHDI	2124.695
yi ~ 1 + effort + RoadDens + S2_sprg_clust1_SHDI	2124.731
yi ~ 1 + S2_sprg_clust1_nclust + S2_sprg_clust1_SHDI	2124.747
yi ~ 1 + S2_sprg_clust1_SHDI + S2_sprg_mosaic3_sd	2124.820
yi ~ 1 + RivDens + S2_sprg_clust1_SHDI	2124.829
yi ~ 1 + effort + RoadDens	2124.870
yi ~ 1 + AspectNort + S2_sprg_clust1_SHDI	2124.956
yi ~ 1 + effort	2124.972
yi ~ 1 + S2_sprg_clust1_SHDI + S2_sprg_mosaic4_sd	2125.084
yi ~ 1 + Altitude + S2_sprg_clust1_SHDI	2125.109
yi ~ 1 + EVI_SPRG_tsMN_sd + S2_sprg_clust1_SHDI	2125.122
yi ~ 1 + EVI_SPRG_tsRG_mean + S2_sprg_clust1_SHDI	2125.159
yi ~ 1 + NumFires_1 + S2_sprg_clust1_SHDI	2125.226
yi ~ 1 + S2_sprg_clust1_SHDI + S2_sprg_mosaic2_sd	2125.228
yi ~ 1 + S2_sprg_mosaic2_mean	2125.236
yi ~ 1 + ViewshedNu + S2_sprg_clust1_SHDI	2125.297
yi ~ 1 + EVI_SPRG_tsRG_sd	2125.307
Summer - M1	
yi ~ 1 + RoadDens	2125.496
yi ~ 1 + RivDens	2126.666
yi ~ 1 + Slope815	2126.708
yi ~ 1 + AspectOest	2126.758
yi ~ 1 + AspectNort	2126.819
yi ~ 1 + effort	2126.886
Summer - M2	
yi ~ 1 + S2_summ_mosaic3_sd	2126.158

yi ~ 1 + EVI_SUMM_tsRG_mean	2126.174
yi ~ 1 + S2_summ_mosaic2_mean	2126.648
yi ~ 1 + S2_summ_mosaic2_sd	2126.701
yi ~ 1 + S2_summ_clust1_SHDI	2126.715
yi ~ 1 + EVI_SUMM_tsRG_sd	2126.813
yi ~ 1 + EVI_SUMM_tsMN_sd	2126.888
Summer - M3	
yi ~ 1 + RoadDens + S2_summ_mosaic3_sd	2125.394
yi ~ 1 + RoadDens	2126.158
yi ~ 1 + S2_summ_mosaic3_sd	2126.174
yi ~ 1 + EVI_SUMM_tsRG_mean	2126.422
yi ~ 1 + S2_summ_mosaic2_mean	2126.648
yi ~ 1 + RivDens	2126.666
yi ~ 1 + S2_summ_mosaic2_sd	2126.701
yi ~ 1 + Slope815	2126.708
yi ~ 1 + S2_summ_clust1_SHDI	2126.715
yi ~ 1 + AspectOest	2126.758
yi ~ 1 + EVI_SUMM_tsRG_sd	2126.813
yi ~ 1 + AspectNort	2126.819
yi ~ 1 + RoadDens + EVI_SUMM_tsRG_mean	2126.829
yi ~ 1 + effort	2126.886
yi ~ 1 + EVI_SUMM_tsMN_sd	2126.888
Winter - M1	
yi ~ 1 + RoadDens + Altitude	2136.612
yi ~ 1 + RoadDens	2137.811
yi ~ 1 + Altitude	2138.160
yi ~ 1 + RoadDens + Slope815	2138.195
yi ~ 1 + RoadDens + RivDens	2138.423
yi ~ 1 + effort + RoadDens	2138.621
yi ~ 1 + RoadDens + AspectNort	2138.637
yi ~ 1 + RivDens	2138.652
yi ~ 1 + RoadDens + AspectOest	2138.673
yi ~ 1 + RoadDens + ViewshedNu	2138.706
Winter - M2	
yi ~ 1 + S2_wint_mosaic2_sd	2137.930
yi ~ 1 + S2_wint_clust1_nclust	2138.559
yi ~ 1 + S2_wint_mosaic3_sd	2138.609
yi ~ 1 + S2_wint_mosaic2_mean	2138.863
yi ~ 1 + EVI_WINT_tsMN_sd	2138.979
yi ~ 1 + S2_wint_clust1_SHDI	2138.992
yi ~ 1 + EVI_WINT_tsMN_mean	2139.141
Winter - M3	

yi ~ 1 + RoadDens + Altitude + S2_wint_mosaic2_sd	2136.712
yi ~ 1 + RoadDens	2137.274
yi ~ 1 + RoadDens + S2_wint_mosaic2_sd	2137.811
yi ~ 1 + Altitude	2137.930
yi ~ 1 + S2_wint_mosaic2_sd	2138.160
yi ~ 1 + RoadDens + Altitude	2138.164
yi ~ 1 + RoadDens + S2_wint_mosaic3_sd	2138.195
yi ~ 1 + RoadDens + Slope815	2138.307
yi ~ 1 + Altitude + S2_wint_mosaic2_sd	2138.344
yi ~ 1 + RoadDens + S2_wint_clust1_nclust	2138.369
yi ~ 1 + RoadDens + S2_wint_mosaic2_mean	2138.423
yi ~ 1 + RoadDens + RivDens	2138.507
yi ~ 1 + RoadDens + S2_wint_clust1_SHDI	2138.559
yi ~ 1 + S2_wint_clust1_nclust	2138.590
yi ~ 1 + RoadDens + EVI_WINT_tsMN_mean	2138.594
yi ~ 1 + RoadDens + EVI_WINT_tsMN_sd	2138.609
yi ~ 1 + S2_wint_mosaic3_sd	2138.617
yi ~ 1 + effort + RoadDens	2138.621
yi ~ 1 + RoadDens + AspectNort	2138.637
yi ~ 1 + RivDens	2138.652
yi ~ 1 + RoadDens + AspectOest	2138.673
yi ~ 1 + RoadDens + ViewshedNu	2138.706

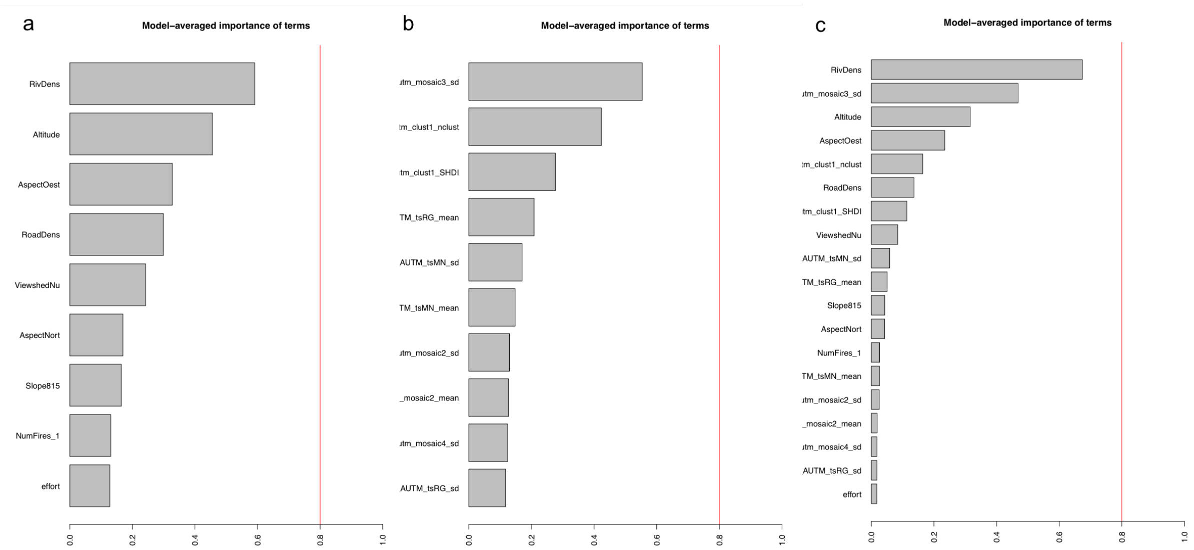

Figure S6.4. Contribution of each predictor for explaining the variation of non-native tree contributions to cultural services in autumn, considering the competing model: M1 (a), M2 (b) or M3 (c). See table C1 for the acronyms of predictors.

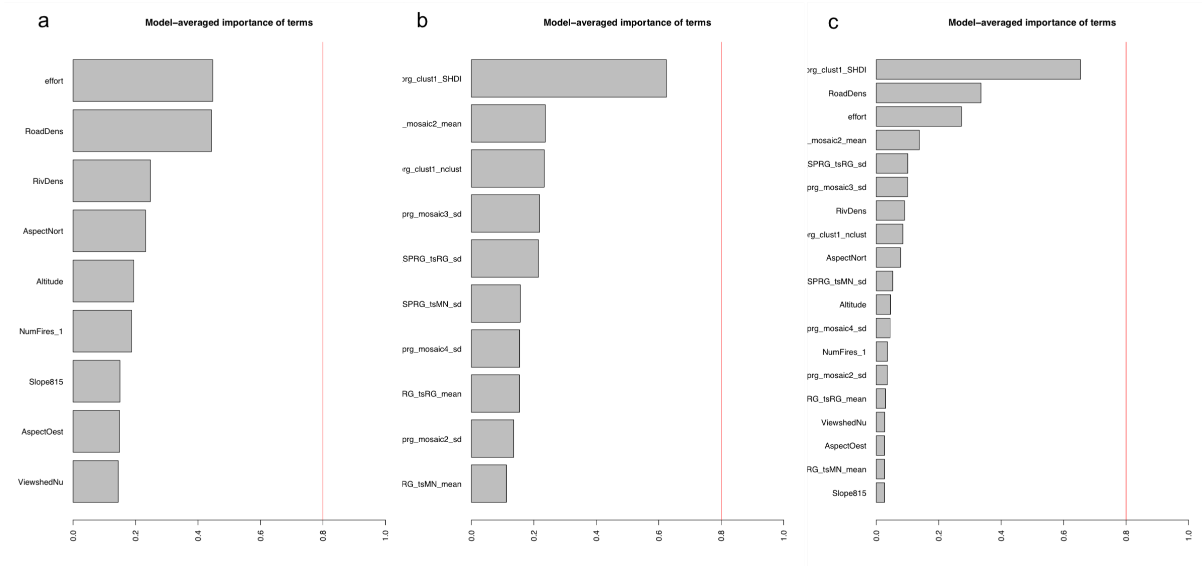

Figure S6.5. Contribution of each predictor for explaining the variation of non-native tree contributions to cultural services in spring, considering the competing model: M1 (a), M2 (b) or M3 (c). See table C1 for the acronyms of predictors.

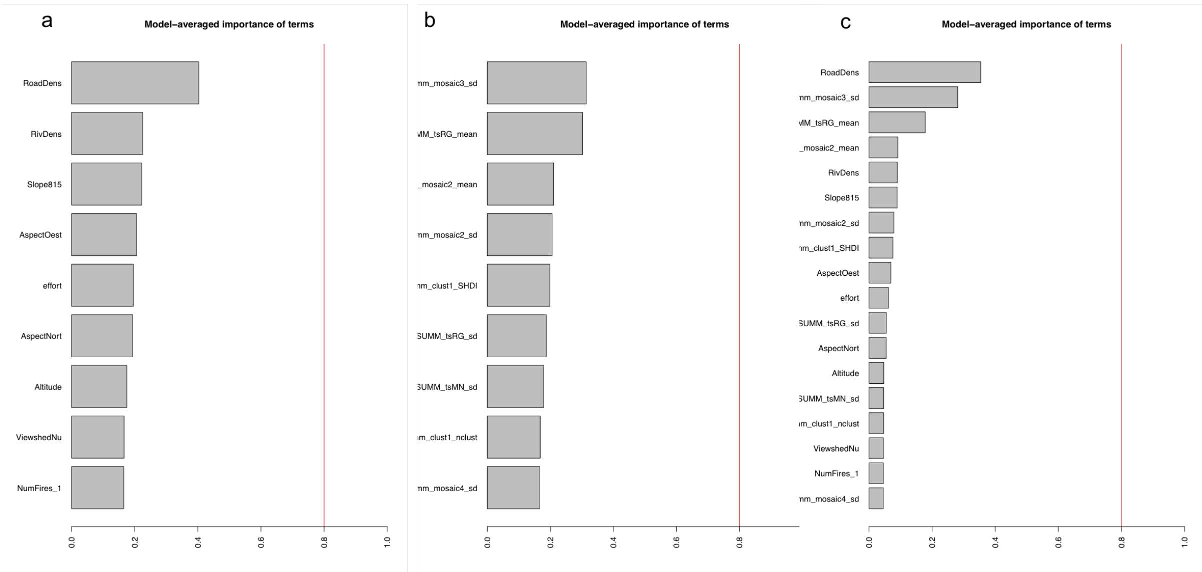

Figure S6.6. Contribution of each predictor for explaining the variation of non-native tree contributions to cultural services in summer, considering the competing model: M1 (a), M2 (b) or M3 (c). See table C1 for the acronyms of predictors.

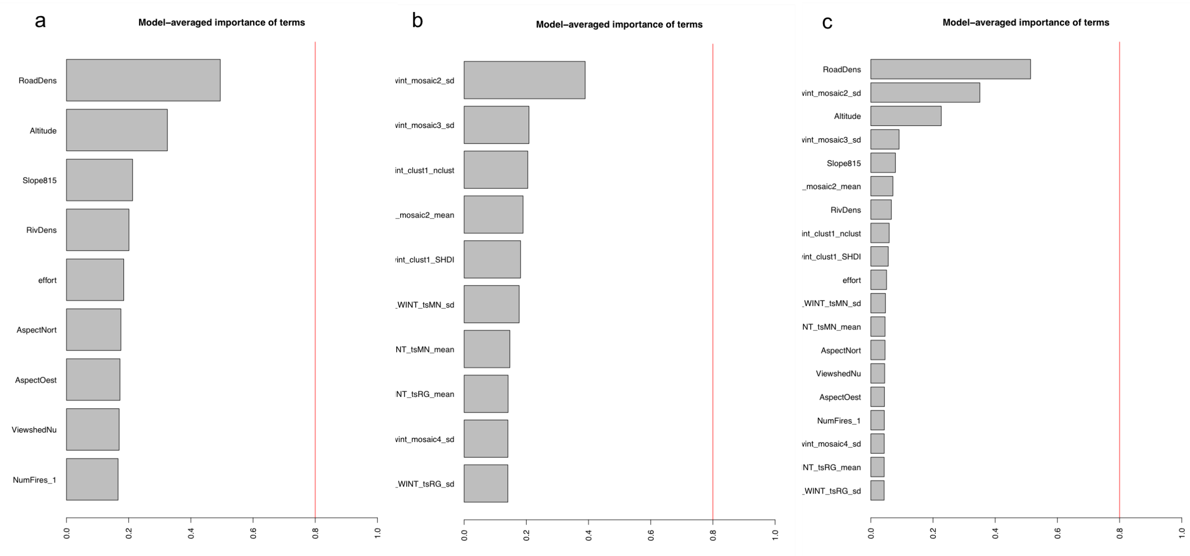

Figure S6.7. Contribution of each predictor for explaining the variation of non-native tree contributions to cultural services in winter, considering the competing model: M1 (a), M2 (b) or M3 (c). See table C1 for the acronyms of predictors.

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Table S6.7. Results from model selection and multi-model framework in the meta-analytical context explaining the odds ratio across the four meteorological seasons. Competing models are presented from the best to least fit hypothesis, determined by AIC values. LogLik: log-likelihood; D2: deviance between observed model and predicted values under a null model; AICc: Akaike information criterion value; ∆AICc: difference between the AICc value of the competing model and the best model in the data set; wi: weight of the model in relation to the whole set of models; VIF: variance inflation factor; QM: heterogeneity of the explained response variable (tested by means of the Q statistics). The table also shows the slope (R2), standard error and statistical significance (under 10000 permutations) of each predictor under the mixed-effects meta-regression model.

	LogLik	D2	AICc	∆AICc	wi	VIF	QM	Predictors	R2	SE	p
Autumn											
M3	-1057.32	0.54	2122.71	0.00	0.62	1.27	5.84				0.01
								River Density	-0.10	0.01	0.01
								Color heterogeneity (green-band)	0.24	0.07	0.04
M1	-1057.78	0.43	2122.78	0.07	0.30	1.49	4.93				0.01
								River Density	-0.10	0.01	0.01
								Altitude	0.04	0.32	0.01
M2	-1058.32	0.34	2123.81	1.10	0.08	1.21	3.84				0.02
								Number of clusters	-0.06	0.05	0.01
								Color heterogeneity (green-band)	0.22	0.07	0.01
Spring											
M3	-1058.01	0.59	2123.35	0.00	0.68	1.23	4.08				0.01
								Shannon diversity of clusters	-0.35	0.02	0.02
								Road Density	0.08	0.07	0.05
M2	-1058.24	0.36	2123.45	0.10	0.23	1.08	3.63				0.02
								Shannon diversity of clusters	-0.36	0.02	0.01
								Color heterogeneity (blue-band)	0.11	0.22	0.01
M1	-1058.40	0.32	2124.62	1.26	0.09	1.26	3.30				0.02
								Accessibility effort	-0.09	0.07	0.01
								Road Density	0.21	0.07	0.04
Summer											
M3	-1059.18	0.25	2125.39	0.00	0.47	1.24	2.57				0.07

								Road Density	0.10	0.07	0.06
								Color heterogeneity (green-band)	-0.15	0.16	0.16
M1	-1059.68	0.15	2125.49	0.10	0.34	-	1.57				0.09
								Road Density	0.09	0.07	0.09
M2	-1060.06	0.08	2126.16	0.77	0.19	-	0.81				0.15
								Color heterogeneity (green-band)	-0.14	0.15	0.15
Winter											
M1	-1065.05	0.67	2136.61	0.00	0.73	1.49	3.13				0.01
								Altitude	0.24	0.03	0.05
								Road Density	-0.10	0.07	0.05
M3	-1064.26	0.40	2136.71	0.10	0.21	1.54	4.71				0.03
								Road Density	-0.10	0.07	0.05
								Color heterogeneity (blue-band)	0.17	0.14	0.05
								Altitude	0.26	0.31	0.05
M2	-1065.95	0.11	2137.93	1.32	0.06	1.49	1.34				0.05
								Color heterogeneity (blue-band)	0.16	0.14	0.05

Appendix I - Details on the Peneda-Gerês National Park

The following figure (Figure S6.8) shows the map of the Peneda-Gerês National Park, which is part of the test area. The original and high-quality map is available through the webpage at: http://www2.icnf.pt/portal/ap/resource/img/pnpg/mapas/map-ingl (accessed 07.05.2018).

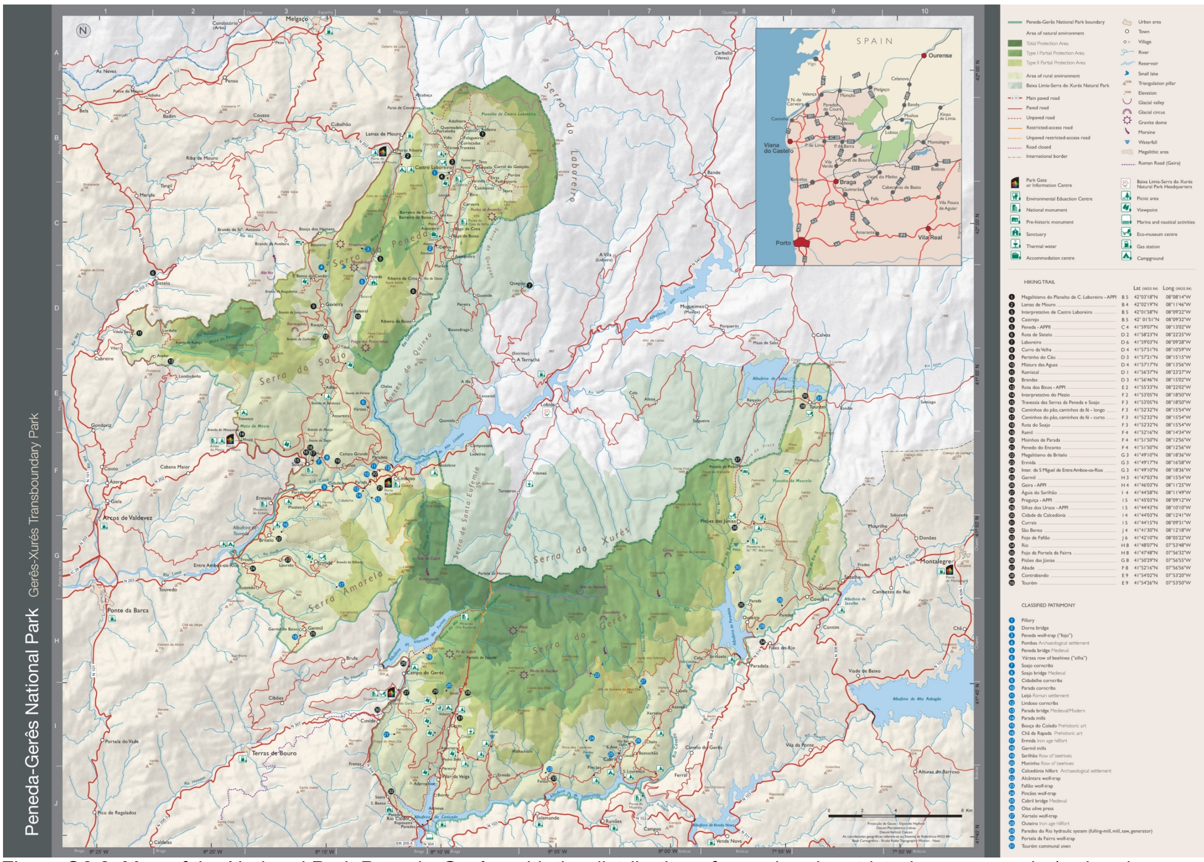
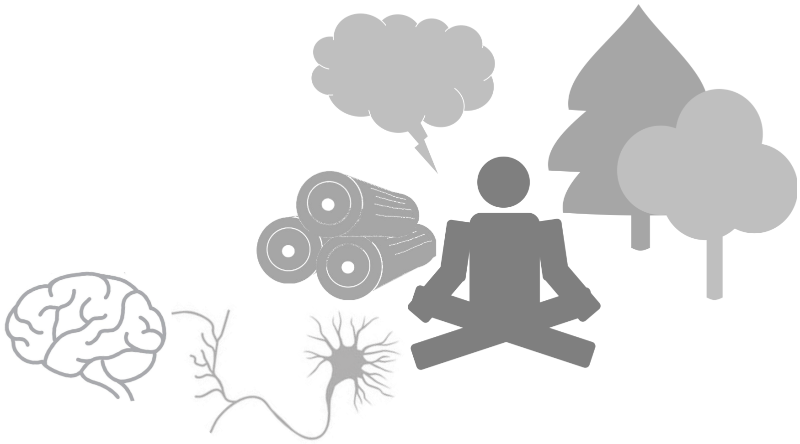


Figure S6.8. Map of the National Park Peneda-Gerês, with the distribution of paved and restricted-access roads (red and orange lines, respectively), water courses (blue lines and polygons), and total (dark green), type I partial (medium green), and type II partial (light green) protection areas. Black symbols represent the main villages in the park, whereas blue symbols refer to the classified patrimony. The park gates (i.e. the main entrances of the park) are indicated with a multi-coloured symbol with a black background.

CHAPTER 7. ECOSYSTEM SERVICES & INVASIONS: THE ROADS AHEAD



DISCLAIMER

This chapter includes two original contributions of this thesis. The first one has been published as a *Letter to the Editor* in journal *Ecosystem Services*, under the title "*Transplanetary" perspective of cultural ecosystem services - Extending Dickinson and Hobbs (2017)'s definitions, characteristics and challenges of cultural services' research*", by:

Ana Sofia Vaz[a,b], Helena Santos[a,b]

[a]Research Network in Biodiversity and Evolutionary Biology, Research Centre in Biodiversity and Genetic Resources (InBIO-CIBIO), Universidade do Porto, Campus Agrário de Vairão, Rua Padre Armando Quintas, PT4485-661 Vairão, Portugal
[b]Faculdade de Ciências, Universidade do Porto, Rua do Campo Alegre, s/n, PT4169-007 Porto, Portugal.

The authors thank João F. Gonçalves (InBIO-CIBIO) for showing some of the illustrative weblinks on "Earth as an art".

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The second one has been sent as a *Letter* and is under consideration in journal *Trends in Ecology and Evolution*, under the title *"Time for remote sensing in invasion science*", by:

Ana Sofia Vaz[1,2,], Joana R. Vicente[1,3], Domingo Alcaraz-Segura[4,5,6], João P. Honrado[1,2]

[1]Research Network in Biodiversity and Evolutionary Biology, Research Centre in Biodiversity and Genetic Resources (InBIO-CIBIO), Universidade do Porto, Campus Agrário de Vairão, Rua Padre Armando Quintas, PT4485-661 Vairão, Portugal;
[2]Faculdade de Ciências, Universidade do Porto, Rua do Campo Alegre, s/n, PT4169-007 Porto, Portugal;
[3]Laboratory of Applied Ecology, CITAB − Centre for the Research and Technology of Agro-Environment and Biological Sciences, University of Trás-os-Montes e Alto Douro, Vila Real, Portugal
[4]Departamento de Botánica, Facultad de Ciencias, Av. Fuentenueva, Universidad de Granada, 18071, Granada, Spain;
[5]iecolab. Interuniversitary Institute for Earth System Research (IISTA) - Universidad de Granada, Av. del Mediterráneo, 18006, Granada, Spain;
[6]Andalusian Center for the Assessment and Monitoring of Global Change (CAESCG) - Universidad de Almería, Crta. San Urbano, 04120, Almería, Spain.

This research contributes to the work done within the GEO BON working group on Ecosystem Services.

Citation: Vaz, A.S., Vicente, J.R., Alcaraz-Segura, Honrado, J.P. Time for remote sensing in invasion science. Trends in Ecology and Evolution. *(Submitted)*

7.1. Remote sensing and ecosystem services: A "transplanetary" perspective

Abstract

In their review, Dickinson and Hobbs (2017) ensemble a set of critiques on the notion, characteristics, and challenges of cultural ecosystem services and respective research. Concurrently, Costanza et al. (2017) provide a throughout overview on the last *"Twenty years of ecosystem services*". Inspired by their narratives, we advance thought on the existence of further challenges in cultural ecosystem services research, relating with the possibility that space missions motivate innovative human-ecosystem interactions, and therefore, novel co-productions of cultural ecosystem services. This is mainly because humans, as observers, are increasingly being shifted beyond the Earth's orbit. Examples include Earth's admiration by people on-board orbital crafts, or through the lens of satellite imagery. This "transplanetary" perspective challenges the conceptualisation and characterisation of cultural ecosystem services. It recognises cultural ecosystem services as co-productions of both humans and the Earth, which can still be immaterial, incommensurable or valued (monetarily or not). It is further characterised by being time-specific, since different time moments in Earth observation can promote distinct human-environment co-productions. Though it is a matter of time for research efforts to expand into the interplanetary dimension of cultural ecosystem services, the roads ahead in this journey might bring exciting research questions.

Keywords: Cultural benefits; Earth observation; Interplanetary perspective; Outer space; Space imagery

Letter to the Editor

In their recent review, Dickinson and Hobbs (2017) put together forthright critiques on the notion and characteristics of cultural ecosystem services, providing a holistic overview based on the recent growth of published research within and outside the academic world. The paper raises valid arguments about cultural ecosystem services, including the multifaceted concepts and typologies, the diversity of models and characteristics, and the challenges that lie ahead for cultural ecosystem services' research. Here, cultural services are defined as the contributions of natural capital, combined ''*with built, human, and social capital, to produce* *recreation, aesthetic, scientific, cultural identity, sense of place, or other 'cultural' benefits*" (Costanza et al., 2017; p. 5).

In this letter, we come to stimulate and incite thought on the notion that the growth of Earth observation and space exploration missions, particularly since the last four decades (Belward and Skøien, 2015), is also motivating novel interactions between humans and nature, and therefore, new human-environment co-productions of cultural ecosystem services (after Dickinson and Hobbs, 2017). Examples include the admiration of planet Earth considering aesthetic, artistic, educational, spiritual, and/or scientific values (following Costanza et al., 2017) either indirectly, through the lens of satellite imagery, or directly by humans, e.g. on-board the International Space Station.

These novel interactions emerge mainly because the scale of observation (i.e. man as an observer), is increasingly being moved beyond the orbit of planet Earth (''transplanetary''). For instance, a tenfold increase per-year in the number of launched Earth observation satellites has been observed from 1970 to 2013 (Belward and Skøien, 2015). Furthermore, the number of expedition crews moving between the *International Space Station and Earth* has duplicated during the last seven years (https://www.nasa.gov/mission_pages/station/expeditions/past.html).

The recognition of this new technological era, including, e.g. the possibility of future missions promoting human space colonization and interplanetary transportation (e.g. SpaceX's transportation systems), poses more challenges for researching cultural ecosystem services (but also other ecosystem services) than those narrated by Dickinson and Hobbs (2017; p.: 185; also, Costanza et al., 2017). Taking inspiration from their narratives, as well as from Costanza et al.'s *''Twenty years of ecosystem services: How far have we come and how far do we still need to go?*" (2017), we briefly present our viewpoint on some of the challenges to the conceptualisation and characterisation of cultural services from an outer space perspective.

Conceptually, by considering Earth observation as a source of ecosystem services it becomes difficult to delimit the ''ecosystem(s)" underlying the services (or the service providing units; Potschin and Haines-Young, 2011), since overall, the Earth is a set of ecosystems functioning and processing in a dynamic way. In this sense, the consideration of Earth as a global ecosystem from an orbital view may not constitute a biophysical problem in itself, but certainly complicates the understanding of the planet as an intricate framework of multiple social-ecological systems (Potschin and Haines-Young, 2011).

An outer space perspective of cultural ecosystem services also makes hard to define the conceptual boundaries between the environmental and the social systems underlying cultural services (after Costanza et al., 2017). This is because Earth's structure, functions and processes will only provide cultural ecosystem services when Earth is interpreted by an out-of-orbit observer. This inevitably resumes interplanetary or orbital cultural ecosystem services as outputs of a co-production between humans and the Earth. For instance, a photograph captured by a satellite[6] or the Earth seen from the window of the *International Space Station*, can potentially be considered a human-environment co-production. The co-produced visual image can be realised as both the result of Earth's dynamic functioning and people's processing of information.

Characteristically, ‘‘transplanetary” cultural ecosystem services can be immaterial or incommensurable (e.g. sense of place from Earth-attachment feeling - i.e., the overview effect or knowledge from the planet itself). Yet, they can nevertheless be valued, as a measure of importance either be monetary (e.g. willingness to pay or actual cost of the satellite imagery for art exhibition purposes) or not (e.g. increased or decreased contribution of Earth observation to human inspiration). Interestingly, a ‘‘transplanetary” perspective of cultural services, would not be characterised by being place-specific (Dickinson and Hobbs, 2017; p.: 184), since there is still not a known way to replicate cultural ecosystem services in other planets with a living component. Instead, orbital cultural services would be strongly time-specific, since a different time moment in Earth observation would promote distinct human-environment co-productions.

We still know very little about the recognition of Earth observation as a source of cultural ecosystem services. Yet, it is a matter of time for research efforts to expand into the interplanetary cultural dimension of ecosystem services. The roads ahead in this journey might bring exciting research questions that are still unresolved within the planetary boundaries. Which interplanetary/orbital cultural ecosystem services are yet to be discovered? How to improve notions and typologies on human-environment cultural co-productions from outer space? Can these notions and typologies be turned into practical usefulness for governance

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Some online sources that use satellite imagery as a source of art (i.e. Earth as a source of art) are: https://twitter.com/S_cartography; https://www.theverge.com/ 2014/4/11/5,603,890/free-satellite-images-from-landsat-7-earth-as-art-series; https://fineartamerica.com/art/satellite+imagery; https://www.nasa.gov/connect/ebooks/earth_art_detail.html; http://www.ustream.tv/channel/live-iss-stream; https://earthnow.usgs.gov/observer/S or well-being (after Potschin and Haines-Young, 2011)? What kind of cultural services valuations are most adequate on a ‘‘transplanetary” perspective? - are just a few questions that may come in the 21st century and defy the ‘‘*new economic paradigm that puts ‘nature’ at the core*” (after Costanza et al., 2017; p.: 13).

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7.2. Remote sensing and biological invasions: An emerging issue

Abstract

Ricciardi et al.'s "*Invasion science: a horizon scan of emerging challenges and opportunities*" propose 14 emerging issues to address biological invasions. Surprisingly, they did not consider the current remote sensing revolution in invasion research. We discuss remote sensing as the 15th issue with emerging challenges and opportunities for invasion science.

Keywords: Biological invasions, Earth observation, management, sensors

Letter

In their review, Ricciardi et al. (2017) identify and discuss key scientific, technological, and socio-political issues expected to drive invasion science and management in the near future. Supported by expert consultation, the authors emphasise 14 emerging issues to address biological invasions, describing their degree of development, influence, impact, and novelty. Surprisingly, their horizon scanning did not identify the current revolution of remote sensing in ecology (Kwok, 2018; He et al., 2015), particularly in invasion science (Juanes et al., 2018; Vaz et al., 2018). Inspired by the authors' view, we propose the consideration of remote sensing as the 15th issue with emerging relevance and novelty for understanding and managing invasions worldwide, and with remarkable opportunities for future development in invasion science.

Ricciardi et al. acknowledge the use of "*high resolution satellite imagery as a tool for monitoring invasions*" as a potential issue of interest (topic 17: Supplemental information in Ricciardi et al., 2017). However, it was placed in a relatively low rank during the horizon scanning (27th position among 40 candidate topics). We advocate that this may have occurred due to the fact that remote sensing represents much more than satellite imagery, serving other relevant purposes besides monitoring, useful for invasion science (Juanes, 2018; Vaz et al., 2018). Remote sensing can be broadly defined as the process of capturing information about an object without contacting it directly. It can be used to gather information about Earth's systems through remote sensors and supplementary surveying techniques (i.e. Earth observation). Remote sensors can be mounted on-board satellites, airplanes, drones, kites, vehicles, tripods or even humans.

Over the last years, remote sensing has developed to strongly improve our understanding of the drivers, processes, patterns and impacts of biological invasions (Vaz et al., 2018). It has been particularly used to identify and quantify animal invaders (Rocchini et al., 2015), map invasive plants and invaded ecosystems (Müllerová et al., 2017), predict the potential distribution of invasive species (He et al., 2015), and assess invasibility and invasion impacts in ecosystems and their services (Hellmann et al., 2017). Remote sensing applications have been rapidly developing from detecting already established invasions, towards predicting new invasion processes and assessing invasion-induced changes (He et al., 2015; Rocchini et al., 2015). As technology evolves, spectral signatures of a growing number of invasive species, at detailed spatial resolutions, are becoming more useful to support preventive and early-response actions at initial invasion stages (Juanes, 2018). Also, with the increasing access to airborne instruments and the development of new Earth observation missions, remote sensing has become essential for tracking invasions, providing multi-temporal and large-scale information (Pettorelli et al., 2014) that is crucial for effective management.

Despite persisting challenges in the remote sensing arena, e.g. related to spatial, temporal and spectral resolution as well as data accessibility and processing, the future of remote sensing will offer even more opportunities to target key challenges in the management of biological invasions. It will refine our capacity to predict, detect and assess invasive species' occurrence and distribution, as well as their impacts on ecosystems' functions and services (Box 5.1). Expected improvements comprise higher availability of multispectral optical imagery with increasing spatial, spectral and temporal resolutions, based on new satellites and sensors from public agencies (e.g. Landsat-8, Sentinel-2, EnMap) and private enterprises (e.g. Digital Globe, Planet Labs, Black Sky or Airbus Defence and Space). These improvements also comprise unmanned aerial vehicles and phenocams with increasing multispectral, hyperspectral and thermal imaging, and LiDAR. In addition, the recent application of machine learning and computer vision methods to high resolution imagery, will open the possibility of detecting and monitoring plant and animal populations with unprecedented efficiency (Guirado et al., 2017; Martinez et al, in press).

Moreover, a large amount of high-resolution data is now available and shared through a revolution of emerging open-source and user-friendly platforms with increasing processing capabilities (e.g. Google Earth Engine, Remap and AppEEARS; Kwok, 2018). Alongside, we are witnessing the integration of remote sensing information in data sources from wider disciplines (e.g. social media, citizen science, molecular information; Kissling et al., 2018), through exceptional computational algorithms and processing approaches (He et al., 2015), such as data-fusion techniques and artificial intelligence (Guirado et al., 2017). These future developments will further widen the horizon for invasion science's remote sensing revolution.

Box 7.1. Remote sensing as the 15th issue in invasion science (after Ricciardi et al., 2017; Vaz et al., 2018).

Opportunities of remote sensing in invasion science:

- Prediction of invasions: remote sensing linked to modelling frameworks allows the prediction of invasive species' distribution with reduced uncertainties, hence supporting prevention and eradication actions at early invasion stages;
- Detection of invasions: remote sensing applied to detect and evaluate the extent of invasions in invaded areas, enabling the support of any management option to mitigate invasions and their impacts;
- Assessment of impacts: remote sensing as a tool for assessing changes in invaded ecosystems and evaluate the consequent impacts, thereby improving mitigation, restoration and adaptation to those changes and impacts.

Challenges ahead on remote sensing applications to invasion science:

- Supporting field work and experimental studies that can improve accuracy in remote sensing and provide deeper knowledge on invaders' phenology and ecosystem dynamics;
- Stimulating the availability of open and free imagery and tagged image datasets, while producing new and free information about invasive species and their impacts;
- Pursuing statistical and computational solutions that convey higher accuracy and transferability of results across spatial and temporal scales.

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CHAPTER 8. DISCUSSION AND CONCLUSIONS

8.1. ALIEN SPECIES AND HUMAN WELL-BEING

8.1.1. Alien species at the social-ecological interface

As described throughout this thesis, particularly in chapters 1 and 2, many alien species have been introduced to new areas in order to provide resources and services that support peoples' well-being. However, in some cases, despite promoting key ecosystem functions that sustain one or more ecosystem services, alien species also change the functioning or quality of other ecosystem functions (Eviner et al., 2012; Gaertner et al., 2009; van Wilgen and Richardson, 2014). They can also create novel ecological processes and conditions that promote ecosystem disservices (Kueffer, 2017; Potgieter et al., 2018; Vaz et al., 2017b). Alien species can thus provide both benefits and nuisances to people, depending, among others, on the particularities of the invasion process and on human exposure and attitudes at particular social-ecological and spatio-temporal contexts (Kueffer, 2013; Pyšek and Richardson, 2010; Shackleton et al., 2014; Shackleton et al., 2018a).

The first three studies presented in this thesis (chapters 2-4) aimed at clarifying whether an ecosystem (dis)services framework, grounded on the benefits and nuisances for human well-being, can improve the understanding and management of biological invasions as a social-ecological phenomenon (research question 1; see section 1.4.1).

Chapter 2 introduced an ecosystem (dis)services framework, grounded on the benefits and nuisances from invasions to human well-being. This framework considered three components of a social-ecological system: the social realm, the ecological realm, and the social-ecological interface. At the social realm, the impacts of alien species on ecosystems can offer benefits (ecosystem services) or nuisances (ecosystem disservices, as well as reduced services) to human well-being, depending on human values, socio-political conditions, perceptions, attitudes, knowledge and ideals (Essl et al., 2017; Kueffer and Kull, 2017; Shackleton et al., 2018a). At the ecological realm, alien species can affect ecosystem attributes, processes and functions, depending on their invasion potential and on the vulnerability of the recipients social and ecological systems (Kueffer, 2013; Pyšek and Richardson, 2010; Shackleton et al., 2014, 2018a,b). These impacts can be translated by an increase of some ecosystem processes (e.g. carbon sequestration; Dickie et al., 2014), by the reduction of other ecological processes (e.g. water availability; Levine et al., 2003; Pyšek and Richardson, 2010), and by new processes or conditions that allow the emergence of ecosystem disservices (e.g. pollen production with allergenic potential; Schindler et al., 2015). There are synergies and trade-offs among the impacts of alien species, and hence benefits and nuisances to people, depending on the temporal and geographical contexts, the interactions with other drivers of environmental change, and the institutional, political and technological dimensions of human agency (Bacher et al., 2018; Essl et al., 2018; Shackleton et al., 2018a; Simberloff, 2015). Building on this framework, chapter 2 proposed a management hierarchy to better target alien species at the ecological realm (e.g. remediation of invaded areas by means of context-appropriate management or technology) and the social realm (e.g. public awareness and creation of social norms, mechanisms and opportunities; Vaz et al., 2017b).

Based on this perspective of invasions as a social-ecological phenomenon, **chapter 3** reviewed the progress of interdisciplinary research in invasion science. Social-ecological perspectives on biological invasions have remarkably increased during the last two decades. The social-ecological view of biological invasions seems to focus mostly on three dimensions. The first dimension includes the role of social factors (e.g. government programs, human beliefs, and socio-economy) on the invasion process (McNeely et al., 2001; Queiroz and Pooley, 2018; Rotherham and Lambert, 2011). The second dimension focuses on the impacts of alien species on humans, including how people perceive, think, feel, know and represent these species according to cultural influences and normative issues (Kueffer and Kull, 2017; Shackleton et al., 2018a; Simberloff et al., 2013; Tassin and Kull, 2015). The third dimension concerns aspects of invasion management, such as participatory approaches, public outreach and articulation with social institutions, frameworks and rules (Estevez et al., 2015; Kull et al., 2011; Marchante and Marchante, 2016). Finally, this chapter called for a higher focus on the *"interlinked social-ecological changes"* promoted by alien species, based on more social-oriented perspectives (e.g. combined insights from public information and scientific evidence). This focus would facilitate the implementation of the management hierarchy proposed in chapter 2, aiming to identify and deliberate invasion outcomes, avoid and mitigate invasion risks, and manage and adapt to opportunities emerging from invasions.

Taking this focus in consideration, **chapter 4** included a case-study illustrating how a social-ecological focus can be used to improve the understanding and management of (currently or potentially invasive) alien species. This chapter evaluated how alien trees affect cultural benefits (i.e. recreation and ecotourism, aesthetics, inspiration and cultural heritage) in the Iberian Peninsula, by using public information from online social media. The different effects found in Portugal and Spain, and across these countries' administrative regions (with distinct socio-economic backgrounds), reinforced the pivotal role of the social realm when addressing alien species (Vaz et al., 2018b). Specifically, the social-ecological approach applied in this chapter suggested that the influence of alien tree species on cultural ecosystem services depends on people's preferences towards their "out-of-normal" features and other appealing visual features (Kueffer and Kull, 2017). This influence also depends on the consideration of alien trees as testimonies of historical and cultural events (such as maritime expeditions; Pooley and Queiroz, 2018), as well as on the level of socio-economic and educational development and human well-being. Considering these findings, chapter 4 also proposed a set of management actions aiming to prevent the naturalisation and spread of alien trees with known (or potential) impacts on ecosystem services in Iberia. Such actions would aim to integrate risk awareness and biosecurity efforts (Marchante and Marchante, 2016) among tourism and ornamental trade entities, and in regions with lower development levels (Andreu et al., 2009; Hulme et al., 2018).

8.1.2. Alien species, invasions, and the rise of remote sensing

How people value alien species and their impacts on human well-being inevitably depends on the changes that those species can induce on the attributes and functions of ecosystems (Larson et al., 2011; Pyšek and Richardson, 2010; Richardson and van Wilgen, 2004; Vilà et al., 2011). These ecological changes vary in space and time, depending on the stage of the (introduction-)naturalisation-invasion continuum (Eviner et al., 2012; Simberloff et al., 2013), as well as on management interventions that promote or hinder such changes (Gaertner et al., 2014; Jeschke et al., 2014; Vilà et al., 2011).

Remote sensing has been increasingly applied to assess and monitor the ecological changes that affect the supply of ecosystem services (Cord et al., 2017; de Araujo Barbosa et al., 2015; Vaz and Santos, 2018) in the light of alien/invasive species (e.g. Dvořák et al., 2015; Müllerová et al., 2017; Müllerová et al., 2013). In this context, **chapter 5** reviewed the progress and extent of the use of remote sensing for addressing alien plants and their impacts on the ecological realm, grounded on the management framework proposed in chapter 2 (research question 2; see section 1.4.1). Remote sensing applications have expanded from detecting already established alien species, towards predicting new invasion processes and assessing the changes they have induced on ecosystems. Overall, contributions from remote sensing were found to be relevant to assess and monitor the whole invasion process, as well as to mitigate, restore and adapt to potential impacts (Simberloff et al., 2013; Vaz et al., 2017b). Remote sensing can be helpful in predictive modelling frameworks, and thus for assisting on the anticipation and early-detection of new invasive species, invaded areas, and impacts

(Große-Stoltenberg et al., 2018; He et al., 2015; Rocchini et al., 2015). The role of remote sensing can further be extended to support prevention and/or eradication actions at early stages of the invasion process (Juanes, 2018; Simberloff et al., 2013), thus contributing to protect ecosystem services and/or avoid potential ecosystem disservices that could result from invasions. Finally, remote sensing can also be used as an effective tool to early-detect potential impacts from invaders, supporting management measures aiming at mitigating and treating impacts on ecosystem properties, functions and services (Bradley, 2014; Dzikiti et al., 2016; Pettorelli et al., 2017).

8.1.3. Combining social-ecological perspectives and remote sensing

Given the increasing importance of social-ecological perspectives and remote sensing in the arena of biological invasions, chapter 5 highlighted the need to combine information from remote sensing with data from social-oriented disciplines, such as social media data (e.g. Kissling et al., 2018; Toth and Jóźków, 2016). By doing so, remote sensing could become more operational when managing alien species and their impacts on ecosystem services (de Araujo Barbosa et al., 2015; Vaz et al., 2018a).

To test the potential of this integrative approach, the case-study presented in **chapter 6** illustrated how social-ecological approaches and remote sensing can be combined to improve the assessment and management of invasions (research question 3; see section 1.4.1). This chapter investigated the contributions of alien trees on cultural benefits from the Peneda-Gerês National Park, in space and time, using social media information together with satellite products and ancillary data. Cultural preferences for alien trees were found to be spatially and temporally dependent. The patterns found in this case-study evidenced that the changes induced by alien trees on cultural ecosystem services are related to their phenology (i.e. at the ecological realm), as well as to their demand for tourism and recreational activities by people (i.e. at the social realm). These results supported the identification of priority areas and time periods (seasons) for adopting biosecurity efforts to prevent and mitigate invasion effects on natural and cultural heritage.

Finally, **chapter 7** discussed possible paths for improving the future assessment and management of invasions, considering the integration of social-ecological approaches and remote sensing advances. Through a couple of stimulating narratives, the chapter discussed the opportunities and challenges in the future research of ecosystem services and biological invasions. The development of new Earth observation missions can motivate novel social-ecological opportunities, and therefore new (human-ecosystem) co-productions of ecosystem services. Remote sensing is also an approach of increasing relevance and novelty for understanding and managing alien species and biological invasions, holding opportunities for the progress of invasion science and management.

8.2. Invasions in the Anthropocene: a dance of aliens, humans and technology

8.2.1. Aliens and humans: a call for interdisciplinarity

As discussed before (chapter 2), the impacts that alien species can produce on ecosystem processes and functions are intrinsically "value-free". Yet, these impacts can be beneficial or detrimental depending on human awareness, perception, vulnerability, attitudes, norms and management, among others (Estevez et al., 2015; Humair et al., 2014; Kueffer and Kull, 2017; Larson, 2005). **Humans influence both the social and the ecological realms, thereby shaping the invasion process**, namely by transferring and introducing alien species, facilitating their establishment and spread, by managing their impacts on the social-ecological system, and by judging their drawbacks and benefits to people (Bacher et al., 2018; Kueffer and Hadorn, 2008; McNeely, 2001; Shackleton et al., 2018b).

The set of studies presented in this thesis showed how **different disciplinary views and scientific methods can be brought together to improve the understanding of biological invasions as a social-ecological phenomenon** (the overarching research goal; see section 1.4.1). This thesis revealed the complexity of synergies and trade-offs among the multiple actors and elements of the social (section 8.1.2) and the ecological (section 8.1.3) realms of invasions. As mentioned above, the changes promoted by alien species on the attributes, functions and processes that underpin ecosystem services and disservices in the ecological context are value-free; yet, the perception and levels of acceptance of benefits and/or nuisances from such services and disservices depend on the social context (see chapter 2). This social context determines: (1) how humans shape the invasion process and facilitate or buffer its impacts on ecosystems; (2) the value of alien species and their impacts on well-being in relation to perceptions, thoughts, emotions and representations; and (3) the management of alien species and their impacts on ecosystem services and disservices in articulation with social actors, frameworks, rules, and behaviours (see chapter 3). These components of the human-alien *dance* are dynamic, inter-related. They depend on the plurality of societal preferences towards particular ecosystem services and also on the specific socio-economic, educational, historical and political contexts that take place at particular geographical and temporal settings (see chapter 4).

Notwithstanding**,** the social realm also responds to the ecosystem attributes and functions which are shaped by alien species. In this regard, **remote sensing technologies, data and methods provide opportunities to assess invasions and their impacts**, at the ecological realm, namely to: (1) detect and evaluate the extent of invasions, and hence impacts on ecosystems and other biota; (2) model and predict the invasion process and the resulting ecological changes; and (3) monitor structural and functional changes in invaded ecosystems (see chapter 5). Inevitably, the type and magnitude of invasion impacts on ecosystem services are complex, depending on the characteristics of alien species, their invasive potential, their distribution and residence time, and on the structural, functional and compositional features of the invaded environment, among other factors (see chapter 6). Since remote sensing can also inform on human presence (potential demand) and land use (management), further Earth observation missions and remote sensing advances will aid the future research of invasions as a social-ecological phenomenon (see chapter 7).

To better address alien and invasive species as a social-ecological phenomenon, insights from different disciplines are required (Kueffer, 2013; Kueffer and Hadorn, 2008; Matzek et al., 2014; Rotherham and Lambert, 2011). Interdisciplinarity has been encouraged at the interface of ecological and social sciences to better comprehend the human dimension that determines and is determined by invasions (Estevez et al., 2015; Vaz et al., 2017a). It has also been motivated by the search for technological solutions, such as remote sensing and habitat modelling, that identify and assess invasion impacts on ecosystem attributes, functions and services (Rocchini et al., 2015; Vaz et al., 2018a). Joint efforts from these distinct views can support management options which inform on deliberating risks and opportunities from alien species to human well-being, under minimum management conflicts (Essl et al., 2017; Estevez et al., 2015; Shackleton et al., 2018a). Concurrently, they would support the monitoring of invasion processes through remote sensing across multiple spatial and temporal scales, and hence ecological changes that underlie the contributions that people obtain from nature (Müllerová et al., 2017; Ricciardi et al., 2017; Vaz et al., 2018a).

It was interesting to note that, despite increasing cross-collaboration among different disciplines, achieving interdisciplinarity in invasion science remains a challenge (see chapter 3; also Abrahams et al., 2018). **To facilitate the integration of social-ecological perspectives and of remote sensing approaches, this thesis incited the rethought of problems, methods, and applications in invasion science** (Hattingh, 2011; Kueffer and Hadorn, 2008; Larson, 2007). Specifically, the focus may need to move from merely identifying "*the invasion by a given species in a particular ecosystem*", to predicting and assessing "*interlinked invasion impacts and social-ecological changes in a given region*" (see chapters 3 and 5). This would still allow a focus on invasive species and invaded habitats, while effectively integrating research cultures, questions and methods from social and technological perspectives.

Ecosystem services offer a good arena for merging insights from science, politics and society to understand, deliberate, mitigate, manage, and adapt to biological invasions. Concurrently, they provide an opportunity to better navigate and understand the many shades of grey in ecosystem dynamics, focusing on the most relevant traits, populations or communities of alien species, from a functional perspective (Abelleira Martínez et al., 2016; Andrew et al., 2014), as opposed to single species and their *"good versus bad"* dichotomy (Larson, 2007). Therefore, managing invasions as a social-ecological phenomenon could: (1) identify which ecosystem functions are being modified by aliens and humans, at which extent and irreversibility level; (2) interpret ecosystem complexity, and the ecological dynamics and feedbacks that can be altered by aliens and humans; (3) recognise opportunities in the modification of ecosystem services to better balance benefits and nuisances associated to alien species, considering distinct measures of human valuation; and (4) account for multiple social-ecological dimensions of change, associated e.g. to the various temporal and spatial scales as well as levels of human intervention and adaptive capacity (see chapter 2).

8.2.2. Managing alien species and invasions in the Anthropocene

This thesis was also able to outline general guidelines to improve the management of biological invasions as a social-ecological phenomenon (see chapter 2).

The *first general guideline* comprises the **identification of potential changes caused by alien species in the ecological realm, including synergies and trade-offs within and among benefits and nuisances from invasions at specific social-ecological contexts**.

Remote sensing has become more accurate in detecting and identifying alien species and invaded ecosystems (see chapters 5 and 7; also e.g. Dvořák et al., 2015; Vaz et al., 2018a). Nevertheless, balancing the benefits and nuisances from alien species to humans, should be considered based on the human preferences, values and interests involved when deciding which management actions are to be implemented, e.g. through deliberation about conflicting views and priorities in invasion management (see chapters 3-6; also Bach and Larson, 2017; Humair et al., 2014).

The *second guideline* involves **prevention and early-detection actions, namely through the protection of ecosystem processes and the avoidance of potential nuisances derived from alien/invasive plants**. Biosecurity actions focused on awareness and education of the notion of “invasive”, “alien” or “non-native” can be useful for preventing risks associated to the trade of alien species with ornamental and market value (see chapters 4 and 6). Also, complementary remote sensing products can support the anticipation, early-detection and prediction of invasive species, invaded areas and impacts on ecosystem services, particularly when applied in modelling frameworks (see chapters 5 and 7; also Große-Stoltenberg et al., 2018; He et al., 2015; Juanes, 2018).

The *third guideline* focuses on **the rapid response to invasions and the mitigation of undesired ecosystem changes, hence allowing to maximise ecosystem services and minimise ecosystem disservices**. Remote sensing can be used to early-detect and evaluate the extent of invasions and their impacts, supporting management measures aiming at mitigating and treating the impacts of these species on ecosystem functioning and services (Bradley, 2014; Cord et al., 2017). This can be translated by distinct actions aiming to consider the invader and its effects e.g. through eradication, containment, or habitat restoration and rehabilitation. Also, the potential of remote sensing to evaluate changes in essential ecosystem functional variables linked to species distributions (He et al., 2015; Rocchini et al., 2015; Vicente et al., 2016) makes it a valuable tool for assessing biophysical changes driven by aliens (see chapters 6 and 7).

Finally, the *fourth guideline* involves **adaptation to the occurrence or expansion of invasive plants, by recognising novel ecosystem services (benefits) or accepting transformations in ecosystem services and the emergence of disservices (nuisances)**. Examples of adaptation include the use of plant invaders for recreational purposes or landscape aesthetics (see chapter 6; also Dickie et al., 2014; Kueffer and Kull, 2017), depending on people’s perceptions, cultural influences, normative issues, and preferences and actions by societal factors (see chapters 3 and 4). Nevertheless, these contributions must consider the impacts of alien species on other ecosystem services (see chapters 2 and 4). Remote sensing can assist on the monitoring of invasions and of their impacts, providing information to adapt to their potential impacts (see chapter 7; also Müllerová et al., 2017; Müllerová et al., 2013).

Nevertheless, **management actions need to consider the particularities of the geographic locations and time periods, since the balance of ecosystem services and disservices will differ across spatial, temporal, social, economic, cultural, and political dimensions**. As exemplified by the case-studies in chapters 4 and 6, the cultural value of alien trees can differ across societal actors, spatial contexts (i.e. countries, administrative regions and protected areas), socio-economic backgrounds (e.g. educational and income levels), and temporal scales (i.e. meteorological seasons). The integration of real human-environment interactions could provide important opportunities for deliberating and outlining solutions based on multiple (actor) interests and uncertainties when dealing with invasions (Head et al., 2015; Kueffer and Hadorn, 2008; Kull et al., 2011; Matzek et al., 2014). Framing invasions from a more balanced social-ecological perspective would help to, among other things, clarify distinct viewpoints relating to perceptions of risks and opportunities, and would help in decision-making by applying collaborative and participatory approaches (Courchamp et al., 2017; Estevez et al., 2015; Heger et al., 2013; Tassin and Kull, 2015).

8.3. Conclusions

The large-scale redistribution of species worldwide by humans constitutes a key fingerprint of the Anthropocene (Head et al., 2015; Kueffer, 2017). On the one hand, humans influence the processes that shape biological invasions, by introducing species to new areas, by facilitating their establishment and by managing ecosystems in ways that enable or hinder the spread of these species (Hui and Richardson, 2017; McNeely, 2001; Vilà and Hulme, 2017). On the other hand, the establishment and spread of alien species brings inevitable consequences for humans, by altering the functions that support ecosystem services and disservices underlying the benefits and nuisances that people use, value and perceive (Bacher et al., 2018; Eviner et al., 2012; Vaz et al., 2017b; Vilà and Hulme, 2017).

This thesis aimed *to improve the understanding and management of biological invasions as a social-ecological phenomenon*, grounded on the integration of social-ecological perspectives and remote sensing advances. The results from the research developed in this thesis were presented in seven studies, arranged in six core chapters (2-7), and previously discussed in chapter 8. Below, the key messages and conclusions from the different chapters are briefly highlighted considering the three research questions proposed in the Introduction.

1. Can an ecosystem (dis)services framework, grounded on the benefits and nuisances for human well-being, improve the understanding and management of biological invasions as a social-ecological phenomenon?

- The ecosystem (dis)services framework proposed in this thesis shows that the changes promoted by alien species on the processes and functions of the ecological realm are intrinsically value-free (i.e. they are what they are, regardless of an a-posteriori valuation). Yet, **the benefits or nuisances deriving from such changes are defined by human perception and levels of acceptance toward invasions, depending on human valuations at the social realm**. Understanding invasions and managing invasions thus need to be considered at the social-ecological interface.
- **Social-ecological perspectives can aid in clarifying how humans value alien species and their impacts on ecosystem services** in relation to perceptions, thoughts, emotions and representations. They can also elucidate how humans shape the invasion process and facilitate or hinder their impacts on ecosystems. These

insights can support a range of management actions targeting alien species, by deliberating risks and opportunities of their effects on human well-being.

- **Management actions targeting alien species need to be tailored for particular geographical, temporal and socio-cultural settings**, since the social-ecological role of alien (and invasive) species also differs across geographic, temporal, social, economic, cultural, and political dimensions. It depends as well on the plurality of societal preferences towards particular ecosystem (dis)services according to specific educational, historical and political contexts, and across a range of scales and policy levels, from broad regions of the globe to countries and their administrative regions.

2. How have invasion research and management taken advantage of the opportunities provided by remote sensing advances, and how can they further benefit from those opportunities?

- **Notwithstanding the importance of the human dimension of invasions, alien species inevitably shape ecosystem attributes, processes and functions, at the ecological realm.** Remote sensing shows promising opportunities to evaluate the drivers, processes, patterns and impacts of invasions.

- **Remote sensing offers a broad range of opportunities for basic and applied invasion science.** It has been extensively applied to detect invasions, from the discrimination of phenological signatures to the identification of dominant or already established invasive populations. When coupled with modelling tools, remote sensing information can also help managers to predict invasion occurrence and spread. Remote sensing can further support the assessment of invasions in a monitoring context, by evaluating changes in the invasion process through space and time.

- **The current and future opportunities provided by remote sensing can make invasion management more operational.** The predictive capacity of models complemented by remote sensing methods and information can support effective prevention and eradication actions to protect ecosystem services and avoid ecosystem disservices. Remote detection of invasions in introduced areas can help in mitigating and treating invasions and their impacts. Also, remote assessments can allow monitoring invasions and mitigating, restoring and adapting to the resulting changes in ecosystem functions.

3. Can social-ecological approaches and remote sensing be combined in integrative frameworks that effectively improve the future assessment and management of invasions?

- The integration of information from different remote sensors and their combination with data sources from other disciplines (e.g. social media, citizen science), and with novel computer processing platforms and algorithms, has shown **potential for improving interdisciplinary assessment and management of invasions and their impacts**.
- **The emergence of computational approaches in social sciences, alongside the current remote sensing revolution, constitutes a significant opportunity to compile and analyse people's experiences and interactions with ecosystems** and hence advancing knowledge about ecosystem services. Concurrently, remote sensing provides spatially- and temporally-explicit information on the functional, visual and sensorial characteristics of ecosystems which underlie how ecosystem services are perceived and moulded by humans.
- Ways forward in the research of ecosystem services and in the science of biological invasions will thus benefit from more combined research and interdisciplinary efforts. This will allow **to focus on invasive species and invaded habitats, while integrating research cultures, questions and methods from social sciences and technological disciplines** to understand the social-ecological impacts of aliens (and invasive) species.

The set of studies presented in this thesis highlighted the high potential for improving the understanding and management of biological invasions as a social-ecological phenomenon. In a time of fast and changing ideas, concepts and approaches, this thesis provides *drops of water to the big oceans* of invasion science and ecosystem services research, perhaps leaving more questions than answers. Yet, it attempts to contribute to the social-ecological course of alien invasions from the perspective of ecosystem services. To advance invasion science from an ecosystem services perspective, many possible steps can be taken in an interdisciplinary sphere. Bringing together different disciplinary backgrounds could create common strategies for targeting the research and sustainability of ecosystem services, challenging, creating and/or extending conceptual questions, research problems, and management approaches associated to invasions. It could **design arenas for advancing the thinking on ecosystem (dis)services, by acknowledging the pivotal role of human agency, technology and management of invasions at the social-ecological interface** in a changing Anthropocene.

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APPENDIX. LIST OF PUBLICATIONS

Here, it is shown the list of publications that I led or participated as co-author, and that were published or are currently under consideration for publication. These publications are within the scope of this thesis. Publications highlighted in bold constitute some chapters of this thesis.

PAPERS IN INTERNATIONAL AND PEER REVIEWED JOURNALS

Vaz, A.S., Vicente, J.R., Honrado, J.P., Time for remote sensing in invasion science. Trends in Ecology and Evolution. *(Submitted)*

Vicente, J.R., Kueffer, C., Richardson, D.M., **Vaz, A.S.**, Cabral, J., Hui, C., Araújo, M., Kühn, I., Kull, C., Verburg, P., Marchante, E., Honrado, J.P. Different environmental drivers of alien tree invasion affect different life-stages and operate at different spatial scales. Forest Ecology and Management. *(Under review)*

Vaz, A.S., Gonçalves, J., Pereira, P., Santarém, F., Vicente, J.R., Honrado, J.P. Earth observation and social media: evaluating the spatiotemporal contribution of non-native trees to cultural ecosystem services. Remote Sensing of Environment. *(Under review)*

Vaz, A.S., Lomba, A., Honrado, J. P., Replacement of pine by eucalypt plantations: effects on the diversity and structure of tree assemblages and implications for landscape management. Landscape and Urban Planning. *(Under review)*

Castro-Diez, P., **Vaz, A.S.**, Silva, J., Van Loo, M., Alonso, A., Aponte, C., Bayón, A., Bellingham, P., Chiuffo, M., DiManno et al. Global effects of non-native tree species on multiple ecosystem services. Biological Reviews. *(Under review)*

Vaz, A.S., Crouzat, E., Santarém, F., Grescho, V., Carvalho-Santos, C. 2018. From pork to fork: toward the social experience of bundles of ecosystem services through gastronomy. Ecosystem Services 32B, 70-72.

Shackleton, R.T., Richardson, D.M., Shackleton, C.M., Bennett, B., Crowley, S.L., Dehnen-Schmutz, K., Estévez, R.A., Fischer, A., Kueffer, C., Kull, C.A., Marchante, E., Novoa, A., Potgiete, L.J., Vaas, J., **Vaz, A.S.**, Larson, B.M.H. 2018. Explaining people's perceptions of invasive alien species: A conceptual framework. Doi: 10.1016/j.jenvman.2018.04.045 *(In press)*

Vaz, A.S., Alcaraz-Segura, D., Campos, J.C., Vicente, J.R., Honrado, J.P. 2018. Managing plant invasions through the lens of remote sensing: a review of progress and the way forward. Science of the Total Environment 642, 1328-1339.

Vaz, A.S., Santos H. 2018. "Transplanetary" perspective of cultural ecosystem services - Extending Dickinson and Hobbs (2017)'s definitions, characteristics and challenges of cultural services' research. Ecosystem Services 29A, 168-169.

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